Available online at www.sciencedirect.com
SCIENCE DIRECT®

Agriculture, Ecosystems and Environment 104 (2004) 5–18

Agriculture Ecosystems & Environment
www.elsevier.com/locate/agee

Policy analysis and environmental problems at different scales: asking the right questions

Thomas P. Tomich[a,∗], Kenneth Chomitz[b], Hermi Francisco[c], Anne-Marie N. Izac[a], Daniel Murdiyarso[d], Blake D. Ratner[e], David E. Thomas[f], Meine van Noordwijk[g]

[a] *ICRAF, PO Box 30677, Nairobi, Kenya*
[b] *Development Research Group, World Bank, 1818 H Street, Washington, DC, USA*
[c] *Department of Economics, College of Economics and Management, University of Philippines, Los Baños, Philippines*
[d] *Department of Geophysics and Meteorology, Bogor Agricultural University, Bogor 16143, Indonesia*
[e] *Institute for Social, Economic, and Ecological Sustainability (ISEES), University of Minnesota, 1985 Buford Avenue, St. Paul, MN 55108, USA*
[f] *ICRAF Chiang Mai, PO Box 267, CMU Post Office, Chiang Mai 50202, Thailand*
[g] *ICRAF SE Asia, PO Box 161, Bogor 16001, Indonesia*

Abstract

In this volume, we seek a common understanding of three environmental problems linked to land use change in Southeast Asia: smoke pollution, degradation of biodiversity functions, and degradation of watershed functions. The objectives of this special issue are to identify usable data and methods for quantifying the impact of land use change on these environmental problems, to identify gaps in either data or methods and, where gaps exist, to set priorities for filling them. That assessment will be done in greater detail in the concluding chapter (Tomich et al., this issue). In this paper, we begin the process by raising policy analysts' basic questions for each environmental problem in turn and making a preliminary assessment of where each of these three problems lies in the 'environmental issue cycle'.

Keywords: Land use change; Environmental services; Missing middle; Smoke; Biodiversity; Watersheds; Environmental policy cycle; Southeast Asia

1. Introduction

Plausible (albeit dire) scenarios for the future in Southeast Asia include increasing conflict over land and water resources and degradation of hydrological, ecological, and other environmental services, which could undermine the stability of national economies, urban centers, and national food security. But do we really know enough about these complex relationships to build a consensus for action? What scientific evidence is available to answer environmental policy questions? Are scientists even asking the right questions? From a policy perspective, Tomich et al. (1999) identified at least three types of questions as crucial:

- *Question Type* 1: Who cares? How are people affected? Are the effects big?

∗ Corresponding author.
Tel.: +254-20-524139/+1-650-833-6645;
fax: +254-20-524001/+1-650-833-6646.
E-mail address: t.tomich@cgiar.org (T.P. Tomich).

0167-8809/$ – see front matter © 2004 Elsevier B.V. All rights reserved.
doi:10.1016/j.agee.2004.01.003

- *Question Type* 2: So what? Is it a policy problem? Would action serve one or more public policy objectives?
- *Question Type* 3: What can be done? Will it work? What are the risks? What will it cost?

These three basic types of policy questions are elaborated below and applied to each of three 'meso-level' environmental concerns: smoke, biodiversity loss, and degradation of watershed functions. A seven-stage 'environmental issue cycle' is presented as a framework for analysis of how the data needs and uses may change with evolution of understanding of a policy problem.

1.1. Environmental insecurity in Southeast Asia

The summary report of the World Commission on Forests and Sustainable Development (WCFSD) speculates that deforestation '. . . could change the very character of the planet and of the human enterprise within a few years . . . ' (Krishnaswamy and Hanson, 1999, p. 6). The press release announcing the WCFSD report included the following statement from George Woodwell of the Woods Hole Research Center: '. . . Forests have a role in supplying the world with timber and fiber. . . . But while those products can be partly substituted, the forests' ecological services for a functioning world cannot' (Lalley and Magnino, 1999). These statements reflect relatively recent concern with global environmental issues (climate change, mass extinctions), but they also build on a longstanding literature tying the condition of soil, water, and forest resources to social and economic stability at the regional and national scale (e.g., Carter and Dale, 1974). Such concerns have had particular force in Southeast Asia since the monetary and financial crisis of the late 1990s. Actual effects have been mixed, however. Currency collapses boosted incentives for forest conversion and intensification of natural resource exploitation for exports, possibly contributing to long-term natural resource management problems. But local effects varied, in part because of the parallel contraction in infrastructure investment.

The possibility that land use change and natural resource degradation could disrupt the economic and social basis of Southeast Asian nations seems plausible enough. For many countries in the region, irrigated rice production in the lowlands is the foundation of national food security. High population densities in rural areas and (until the interruption in the late 1990s) rapid growth in urban industry and services each contributed environmental pressures. But how much do we really know about relationships between land use change and the environmental services on which national economies and local livelihoods depend? 'Natural capital' is economists' jargon for the stocks of natural resources (including soil, water, air, vegetation, wildlife, and other organisms) and for the interactions among these that supply environmental services (Costanza et al., 1997; Izac, 1997). Table 1 lists some examples of the wide range of environmental services at different scales that may be affected by land use change. Many of these cut across scales, such as the supply of raw materials (e.g., food, fodder, fiber, medicines, resins, timber) and the moral value of preventing extinctions. Although 'environmental services' often have been treated as synonymous with 'forest functions,' we prefer the former term because even if forest-derived land uses are not perfect substitutes for natural forests, they still provide some level of these services.

Table 1 also could include a large number of environmental services (and disservices) directly affecting human health, which of course are crucial to human welfare. Land use change per se (see Roulet et al., 1998) and all of the major themes explored in the balance of this paper—smoke, biodiversity, watersheds—have major public health implications. The literature on pesticide runoff alone is substantial (e.g. Rola and Pingali, 1993). Many of these concerns are the topic of a recent review of environmental change and human health (WRI et al., 1998). Moreover, it is possible to treat human health as a separate dimension of overall sustainability—as long as human health is reintegrated into the analysis of tradeoffs with production and other environmental effects at some point (Crissman et al., 1998). Although we will mention them briefly below, human public health concerns are omitted from most of this paper.

The global ASB research programme already has made contributions to clarification of tradeoffs between welfare of poor rural households and global environmental services (for Indonesia, see Tomich et al., 1998a, 2001). However, the hydrological, ecological and other environmental services at the local

Table 1
Examples of environmental goods and services at different scales

	Macro	Meso	Micro
Scale	Global	Regional transboundary[a] / National[b] / Local type II: inter-community[c]	Local type I: intra-community[d]
Commodities	Supply of raw materials Scientific and educational materials Options for new and improved raw materials	Supply of raw materials Livelihoods and employment opportunities Cultural, scientific and educational materials Options for new/improved raw materials	Supply of raw materials Livelihoods and employment opportunities Cultural and educational materials
Amenities and protective functions	Climate stability Evolutionary potential for adaptation Cultural, scientific and educational opportunities	Air quality (smoke) Biodiversity functions: pollination, seed sources, seed dispersal, biological pest control, production stability Evolutionary potential for adaptation Water quantity: buffering flooding and base flow Water quality: filtering sediments, decomposing wastes, and diluting other pollutants Aesthetics: values for residents and as basis for tourism.	Nutrient cycling Filtering sediments and water pollutants Microclimate effect of trees Aesthetics: values for residents and as basis for tourism
Moral values	Existence of species Cultural survival/support for livelihoods of indigenous cultures Bequest values of climate stability, biodiversity, and other natural amenities for future generations	Existence of species Cultural survival/support for livelihoods of indigenous cultures Bequest values of biodiversity and other natural amenities for future generations	Existence of species Bequest values of biodiversity and other natural amenities for future generations

Sources: typology of goods and services is adapted from Norton (1988). Other references: Barbier (1995), Brenner (1996), Costanza et al. (1997), Daily (1997), Gowdy (1997), Menz et al. (1997), Pimentel and Wightman (1999), Randall (1988).

[a] Regional transboundary scale environmental effects cross the borders of neighboring countries within a region, such as Southeast Asia.

[b] National scale environmental effects loom large within national borders.

[c] Local Type II: Inter-community environmental effects are landscape or watershed scale effects that span more than one settlement or village, such as the effects of land cover change upstream on hydrology downstream.

[d] Local Type I: Intra-community environmental effects are confined to a single settlement or village.

and national level are a significant gap in this analysis in terms of their impact on local people but also regarding potential complementarity with global environmental objectives. For Southeast Asia, smoke pollution ('transboundary haze'), the functional roles of biodiversity, and watershed functions all fall in this 'missing middle', the gap between local interests and global environmental concerns. The focus here is on meso-level environmental externalities that involve groups and spatial or time scales that are too big for individuals to resolve but that fall within the jurisdiction of a single (or a few) government entities. This underlies the distinction in Table 1 between 'Local Type I' (*intra*-community effects) and 'Local Type II' (*inter*-community effects) and is why the latter are classed as meso- rather than micro-issues. Individuals and small groups may be able to deal effectively with intra-community opportunities and problems on their own, but (like global, transboundary, and national issues), some intervention by a higher authority may be necessary to address inter-community environmental conflicts or to seize opportunities that span multiple communities.

There are several areas of potential conflict between the welfare of households in Southeast Asia's uplands—particularly their pursuit of profitable land use options—and their neighbors downstream (or downwind). Among these perhaps the most pertinent question for the people of Southeast Asia is whether pursuit of profitable land uses undermines key environmental services—translating, for example, into more frequent and more damaging floods, water shortages, and pest outbreaks. The recurrent transboundary smoke problem in Southeast Asia is linked to El Niño, but also is driven by land use change promoted as part of development strategy and resulting conflicts over land. Without interventions to strengthen or create mechanisms for conflict management, the future may bring intensification of social conflicts over natural resources—particularly land and water.

While some have argued that 'artificial' distinctions between global environmental interests and regional, national, and local concerns impede action (UNDP et al., 1994, p. 5), the tradeoffs among objectives spanning these scales should not be ignored. Pursuing global interests in conservation of endangered species and unique ecosystems involves a high-opportunity cost for local people because of land scarcity in much of Southeast Asia. Under these circumstances, it is clear that the feasibility of key conservation objectives rests on the ability to stabilize the boundaries of the so-called 'protected' areas through some combination of incentives and enforcement. Again, this requires capacities for conflict management, including a mechanism for compensating local people for foregone opportunities. Here, some of the successful examples of bioprospecting in Central America and wildlife management for ecotourism in Eastern and Southern Africa may hold useful insights for Southeast Asia. If it is not feasible to realign incentives for local communities though such means, it is inevitable that conservation areas will continue to shrink—ultimately to the point that they no longer function. There also may be scope for finding common ground to couple local development initiatives with global interests in carbon sequestration since, if the possibility of global climate change is realized, its local manifestation may accentuate the frequency and scale of floods, droughts, fires, and pest outbreaks (Jepma and Munasinghe, 1998, p. 49).

2. Overarching questions

The WCFSD report and the statement by Woodwell mentioned above are but two examples of myriad well-intentioned messages aimed at policymakers and the public regarding land use change and environmental services. But do we really know enough to build a consensus for action at the local and national level and the scales in between? How big are the effects of land use change (for better or worse) on stability of production systems at these scales? Although it appears that there are no perfect substitutes for natural forests regarding global environmental issues, some derived land uses may provide some of these services (Tomich et al., 2001). How well do these forest-derived land uses substitute for forests from the perspective of local people and national objectives? To what extent does expansion of shifting cultivation and other smallholder land use systems pose a threat to the 'natural capital' of Southeast Asia?

Three types of overarching questions are the focus of this paper.

2.1. Question Type 1: Who cares?

How are *people* affected? How *big* are the effects? Who loses? Does anybody win? Are the negative (or positive) effects big enough to capture the attention of local people or of policymakers? *What is 'big'*? US$ 30 billion is big by virtually any standard. According to the *Economist* (3 April 1999, p. 93) that was the cost of flooding in China in 1998, which was the biggest economic loss caused by a single event that year. That figure was roughly 4% of China's GDP in the mid-1990s, which accounted for about a third of a year's (very rapid) economic growth for the country. (Economic statistics are from World Bank, 1997.)

The main point is that *'big' is relative*. What is big in Brunei may not attract attention in China; what is big in a Chinese village may not be noticed in Beijing. Do we have the methods and data to answer this question for environmental services in a way that is comparable to peoples' and policymakers' other concerns? Measurement is useful—particularly when setting priorities among disparate, competing objectives—but something can be widely considered to be important even if it is not yet quantified (or is not quantifiable). For example, smoke pollution was recognized as a crisis before EEPSEA (1993–1998) figures were available—indeed work to produce those estimates was a response to the importance of the problem. Some have argued (Norton, 1988; Ehrenfeld, 1988) that there are important values of biodiversity that are unquantifiable. If so, how can these be incorporated in the debate?

2.2. Question Type 2: So what? Is it a policy problem?

Policy research aims to sharpen identification of policy problems and to bring analysis to bear in order to enhance options for meeting fundamental policy objectives, including growth with poverty alleviation, food security, and environmental stability. Tree planting, reforestation, and soil conservation are means to ends; they are not policy objectives themselves. Unfortunately, government agencies often do set targets in these terms, which is part of the problem. For example, the Vietnamese Government has announced a goal to reforest 5 million ha of 'degraded' or 'barren' land by 2010. Setting the target in those terms risks diverting line agencies and local authorities from paying adequate attention to impacts of these projects on local livelihoods, sedimentation and flooding downstream, and other fundamental concerns.

2.3. Question Type 3: What can be done?

This question may come in tandem with other questions, such as *Do we really know enough to act?* And, if a particular action is to be taken, *how do we know it will work?* and *what are the risks?* will be asked. A hallmark of policy research is the practical assessment of specific policy instruments, the means of affecting policy objectives in the 'real world'. Examples of policy instruments relevant to land use change include exchange rates and interest rates; price, trade, and marketing policies; laws and regulations affecting access to and transfer of land and other assets; and public expenditures for infrastructure, research, and extension.

To grapple effectively with these three sets of questions, it is necessary to understand the political, administrative, and legal processes of a particular setting and the ways that various interest groups affect these processes.

3. Externalities, institutions, and scale

Many of the amenities and protective functions listed in Table 1 are externalities (see Box 1 for related definitions), including physical phenomena at the meso-scale ranging from smoke produced by land clearing, to biodiversity functions such as pollination, and watershed functions such as buffering base flows and filtering sediments. Understanding these biological and physical consequences—referred to as 'lateral flows' in the next paper in this collection (van Noordwijk et al.)—is essential to formulating sound policy responses or even knowing whether intervention is needed. When these lateral flows affect other peoples' consumption or production opportunities, economists refer to them as externalities.

Existence of externalities is not a sufficient justification for policy intervention, however, since individuals may be able to negotiate a solution even if markets fail to provide one (Coase, 1960; Zilberman and Marra, 1993). Whether or not such solutions are implemented depends on the value of the externality compared to

Box 1. Social costs and scaling in space and time

Why do not individuals take care of environmental problems themselves? Most economists' answers to this question can be subsumed under three broad categories: policy distortions, market imperfections, and market failures. In each case, market incentives that influence people's land use decisions fail to include the full *social costs* (or benefits) of their choices. *Policy distortions* are government mistakes—at least from the point of view of the public interest, if not from the perspective of the private interests of policymakers and bureaucrats. But even if there were no misguided policies, there still would be plenty of work for policy analysts because markets for many (but not all) environmental services either are imperfect or fail completely. *Market imperfections* include the combined effects of uncertainty and irreversibility; regional growth linkages and spillovers; and economies of scale. They also include high transactions costs, which are discussed further below, and factor market imperfections, such as insecure tenure and lack of access to banking services. These classes of market imperfections operating together can create situations where, once an unforeseen threshold is passed, for all practical purposes, there is no going back. *Market failures*, which include *externalities* and *public goods*, are cases where no market price exists. Effects of externalities and public goods may be felt locally, regionally, or globally; in fact they correspond to many of the environmental services listed in Table 1. The term 'externality' refers to the effects of activities by one economic agent on another that are not reflected in market prices. Externalities may have positive or negative effects (or both). 'Public goods' are a specific form of externality. The defining characteristics of public goods are: (1) their use by one person does not prevent full benefits being enjoyed by others and (2) it often is difficult to exclude users, hence it may be excessively costly to charge them. The global environmental services in Table 1—climatic stability and avoidance of extinctions—are global public goods.

the transaction costs—costs of organizing, negotiating, monitoring, and enforcing agreements—involved in a negotiated solution and on the distribution of power among the interested parties. Generally, transaction costs increase with the number of people involved, their dispersion in space, and differences in timing due to lags between causes and effects. The further removed the impacts are in space and time, the more difficult the organizational challenge.

So local environmental externalities concentrated in a small area and involving a few people (who probably know each other and may even be relatives) and for which there are clear and immediate cause and effect relationships, often will not be a policy problem. For example, long-established communities in Indonesia's Outer Islands often have their own well-developed techniques for managing burning and timber felling in order to avoid accidental damage to neighbors' property and widely recognized compensation rules already exist when accidents do happen (H. de Foresta, pers. commun.).

Conversely, transactions costs are likely to be high for global public goods, including climatic stability and avoiding extinctions, since effects are complex and dispersed globally and in which six billion humans share an interest. But, since landscapes provide mixes of multiple services affecting multiple scales simultaneously, what is the scope for better incentives for local resource management also to contribute to solving the larger-scale problems as a byproduct?

4. Stages of the 'environmental issue cycle'

Unless there is some channel to aggregate feedback from those affected through an incentive system or other method of social control, externalities will be ignored by the land user causing them. This applies whether the externalities are positive (environmental services) or negative (pollution). And while better information about causes and effects may be necessary to identify solutions, information alone typically is not sufficient because there often are conflicting interests between producers of externalities and those who experience the external effects.

In the case of degradation of environmental services (a negative externality) resulting from land use change, direct conflict, sometimes violence, among the 'stakeholders' (i.e., the people concerned on all sides) may result unless some form of authority intervenes. These authorities may be officials or policymakers at various levels of the government bureaucracy—where formal responsibilities are established—but they might also be local leaders, elites or other individuals whose influence derives from positions in clan and other customary institutions, social, religious or civil organizations, or non-governmental organizations. These authorities have at least four strategies to choose among in responding to pressure from the various stakeholders:

- do nothing (ignore the issue for as long as possible),
- compensate the suffering groups,
- mitigate degradation by, for example, increasing the filter functions intercepting lateral flows, as discussed by van Noordwijk et al., this issue, or
- prevent (or reduce) degradation by modifying the behavior of land users (the producers of negative externalities) through regulations, market-based instruments (taxes and/or subsidies), other means of social control, such as negative publicity, or some combination of these approaches.

Even with overt conflicts between land users and the injured stakeholders, considerable pressure may be required before authorities shift into action. Moreover, if no clear (formal or informal) authority yet exists—as often is the case regarding environmental externalities in the 'missing middle' scale in developing countries—the threshold for action is even higher since it requires institutional innovation. On the other hand, those who wield power and authority also have their own interests in resource exploitation. Indeed, those elite interests often drive non-sustainable resource use.

The data that will be most effective in eliciting constructive responses from stakeholders and the authorities—hence the most appropriate research methods—depend on where a particular externality is on the seven stages in the 'environmental policy issue life cycle' (Fig. 1, adapted from Winsemius, 1986):

- *Stage* 1: Perception by 'pioneers' (if they are ultimately judged by society to be correct) or 'crackpots' (if they are shown to be wrong) of a particular environmental issue, but no broader awareness either by society at large or by the authorities.

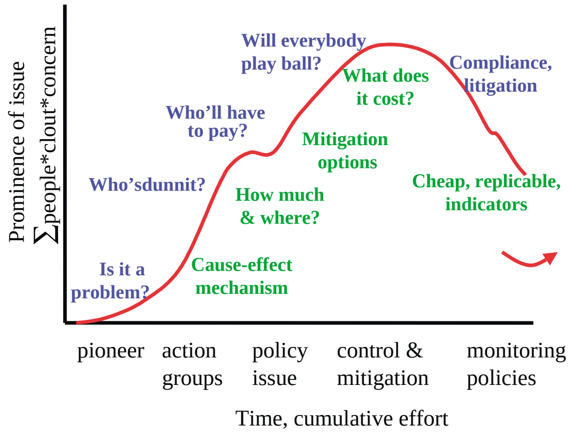

Fig. 1. Schematic 'issue cycle' of an environmental externality in a democracy showing how public perceptions evolve over time through social interaction and scientific enquiry (adapted from Winsemius, 1986, p. 17).

- *Stage* 2: Lobbying by 'action groups', denial of effects by some groups of stakeholders, and incipient awareness but no action by authorities.
- *Stage* 3: Widening acceptance of existence of (potential or actual) environmental impacts, with mounting awareness and pressure for action by authorities.
- *Stage* 4: Debate on evidence of 'cause and effect' and attribution of 'blame'.
- *Stage* 5: Inventory and assessment of prevention and mitigation options and their environmental, economic, and administrative costs and benefits.
- *Stage* 6: Negotiations on prevention or mitigation of impacts.
- *Stage* 7: Implementation, monitoring, and enforcement of prevention or mitigation actions.

The course of these events obviously depends on the particulars of culture, society, polity, and economy. While the concentration of power and decision-making under a centralized, authoritarian regime might appear to accelerate Stages 3–7, there is a greater risk that the process may be 'nipped in the bud' at Stage 2. Broad accountability of the authorities to the public seems to be a decisive element; here the political participation and public debate epitomized by democracy has distinct advantages, at least in the discovery of problems if not necessarily in the identification and implementation of efficient solutions.

Whereas details differ between various environmental issues, the course of events in regional issues such as ‘acid rain’ in NW Europe and global issues such as ‘climate change’ could follow similar paths, at least if actions somehow respond to both the numbers of people concerned and the intensity of their concern (as in Fig. 1). In Stages 2 and 3 of this cycle, research would be needed primarily to test the validity of Stage 1 ‘suspicions’ about a link between an undesirable environmental impact and a change in land use. Establishing a probable cause-and-effect chain as opposed to ‘mere coincidence’ or ‘spurious correlation’ is important at this point as a basis for sound policy intervention. This information also could help to build broader support for action and to undermine resistance from vested interests. In these stages there also is a need to estimate the likely magnitude of impacts—are the effects big or small?—since initial uncertainty may range over several orders of magnitude.

Once awareness and support is formed for action on a particular environmental issue, the debate may shift focus to specification of cause and effect chains, especially where they are important for attribution of blame. Perceived gaps in the quantification of impacts or in causal explanation of the phenomena are major obstacles in Stage 4. This stage also is where positions are staked out for subsequent negotiations among stakeholders. Various stakeholders may agree (Stage 5) on the need for an inventory of prevention and mitigation options; or the process may be more adversarial, with each group applying evidence selectively and advocating a position serving its own interests. If the latter, the negotiation process in Stage 6 and implementation in Stage 7 may require formal mechanisms for dispute resolution and conflict management, either through courts or some other mechanism for arbitration and enforcement. Either way, the outcomes of Stage 6 could be enhanced (from a broad social perspective) from inventory of the impacts and the role of various actors in causing the problem as well as assessment of the various options for action (Stage 5). In Stage 6, simplified parameters (‘rules of thumb’) often are more persuasive than full quantification of webs of cause-and-effect and spillovers in complex models.

If agreement can be reached on some mix of prevention and mitigation, research needs will shift again to those for monitoring either environmental impacts or changes in behavior (or both) and, as necessary, enforcement of pre-agreed sanctions. Standardization of measurement methods is particularly important at this stage. When lack of compliance leads to legal conflicts, the methods for monitoring impacts also are likely to be scrutinized, which could establish a premium on replicability and reliability (low measurement error) and a discount on speculative inquiries into complex causality.

During this progression of stages, social and political processes *ideally* would shift research priorities and methods through a sequence from intensive (process-oriented, cause-and-effect relations, explanatory models) to extensive (spatial databases, long-term monitoring) approaches, with a gradual standardization of measurements and data collection protocols from a (possibly haphazard) pioneering phase. Standardization of methods and general agreement on cause-and-effect chains obviously brings advantages, but it also can become a liability if it prevents critical examination of discordant information and refinement of process-based understanding. At the end of a cycle (Stage 7), or even as early as Stage 4, perceptions of environmental issues may ‘fossilize’ and require either significant time or some disillusioning shock before they are rejuvenated and another round of the cycle ensues. Problems that apparently had been ‘understood’ and ‘solved’ or at least brought under control, may re-emerge in a new cycle if situations change or if the initial diagnosis proves to be incorrect or the interventions ineffective. Once broad support for specific interventions has been built, however, the pioneers for the new cycle may have to come from a different group of stakeholders.

5. Managing smoke

Slashing and burning is the preferred method of land clearing in the tropics—for smallholders and large companies alike—because it is cheap, at least from a private perspective, and relatively easy. In addition, fire eliminates field debris, reduces problems with weeds and other pests and diseases, makes nutrients available in the form of ash (also meaning less reliance on purchased fertilizers) and loosens the soil to make planting easier. In some ways it is preferable environmentally compared to some other land-clearing methods. For example, bulldozers and other heavy machinery cause soil compaction and erosion. But the smoke from these fires also provides a textbook case of divergence between private benefits (cheap, effective land clearing) and social costs (lost opportunities for commerce and tourism from disruption of transport and obliteration of beautiful views, damage to human health (especially asthma and bronchitis), increased absenteeism and reduced worker productivity). In fact, smoke features prominently in Coase's (1960) seminal treatment of externalities.

5.1. Who cares about burning and smoke?

Who benefits most from free use of burning for land clearing, large-scale plantations or smallholders? Which of—or under what circumstances do—these groups contribute the most to smoke problems? How do these costs compare with the direct benefits of burning for land clearing? What are the consequences of land clearing without the use of fire? And who bears the greatest costs of smoke from burning for land clearing? Local people in the neighborhood? . . . people in the province or state? . . . the nation as a whole? . . . people in other countries?

In addition to use of fire as a tool for land clearing, fire also can be a weapon in social conflict (Tomich et al., 1998b). But does arson play a significant role in the smoke problem? Vayda (1998) argues that the incidence of accidental fires may be much higher that is conventionally believed. For a complex situation where firm conclusions are difficult even with detailed case studies (Potter and Lee, 1998), is there any hope of being able to attribute shares of smoke between purposive burning (for land clearing or arson) and accidents? (In addition to the social costs of smoke, simulations by Menz et al. (1997) indicate that risk of fire spreading from neighbors' plots could be a significant disincentive to smallholder tree planting on *Imperata cylindrica* grasslands.)

5.2. What can be done to reduce the smoke problem?

The answers to the 'who cares' questions matter a great deal for the design of interventions—training programmes may be appropriate if most smoke comes from accidental fires, but would be irrelevant or even counterproductive if most fires are set deliberately—but is it feasible to measure these phenomena at a scale relevant for policy formulation? What policy options and policy instruments exist to manage the recurring regional problem of smoke from land clearing? Are there opportunities for action to improve management of smoke through policy reform, institutional strengthening, or public awareness? What are the main lessons from the experience of different countries in designing and implementing strategies to manage smoke? Are there any win–win opportunities? If there are conflicting interests, should/will the victims of smoke compensate people who give up burning? Or should the polluters pay? Is either approach feasible administratively or politically?

5.3. Is burning the problem? Or is it the smoke?

With the return of smoke to the skylines of Singapore and Kuala Lumpur in mid-1999, and with fresh memories of the smoke problems of 1997/1998, ASEAN environmental ministers once again called for immediate implementation of a 'zero-burning' policy (*The Star*, 17 April 1999). Is it possible to go beyond rhetoric and apparently futile past efforts to ban burning to identify more workable options for managing burning to reduce smoke problems? If options exist, who would implement them? Who (or which institution) has the greatest influence over smoke and/or burning for land clearing? How could they influence it? What is a workable unit for management of smoke? ASEAN or other international organizations? The nation? Whole islands? Regions? Specific landscapes? Fields? What role do local ('informal') institutions play in managing burning and smoke?

5.4. What are the priorities for research and for action on burning and smoke?

Given the lack of effective action to date, under what circumstances would more or better data be used? Depending on the weather, the transboundary smoke problem in SE Asia oscillates between Stages 1 and 4 (or even Stage 5) of the 'issue cycle'. Although authorities can ignore the problem between 'crises' there has been mounting public awareness in adversely affected countries and (primarily external) pressure for action by Indonesia, the main source. It remains to be seen whether regional attempts to ignore the problem or to affect the appearance of doing something about it will be accepted in the future, but it is clear that Indonesia's capacity to act was reduced by monetary and political crises in the late 1990s (Makarim, 1999).

At this stage, what data would be most useful in designing and implementing a strategy to manage burning in order to address the smoke problem? Can remote sensing be used reliably and precisely enough to apportion blame? Aside from remote sensing and better understanding of institutional functions at various levels, what other types of data or research would be useful in forging a consensus and identifying viable options for action? Is more or better information the answer?

6. Degradation of biodiversity functions

Much discussion of biodiversity conservation focuses on *global* existence values—in other words, preventing extinctions. Much less attention has been given to *local* functional values of biodiversity (belowground as well as above). Here we seek to put aside, for the moment, legitimate global concerns with extinctions, in order to focus on local, functional roles of biodiversity in landscapes where people seek their livelihoods.

6.1. How does reduction of biodiversity affect peoples' livelihoods?

There is ample evidence that forest conversion reduces biodiversity. The winners are those who profit from forest-derived land uses. But who loses from lower biodiversity richness at the local level? How? Are there threshold effects of biodiversity loss on stability of production such that land use change that could be sustainable for a limited number of 'winners' on a limited area would be an ecological catastrophe if everyone did it? What are the functions of biodiversity in the stability of production systems? How important are these stabilizing functions of biodiversity compared to its other ecological goods and services? For example, are the effects of biodiversity on production stability big or small compared with:

- opportunities for direct use and marketing of forest products by local people, either under normal conditions or during difficult times?
- effects of biodiversity conservation on prevalence of human pests (tigers, elephants) and diseases (malaria)?
- aesthetic and spiritual roles of biodiversity for local people, which also may be developed as a basis for new economic activities such as ecotourism?

6.2. Does biodiversity loss affect national policy objectives?

Should national policy makers worry about loss of biodiversity in the same way they seem concerned about degradation of watershed functions? From a national perspective, how important are the stabilizing functions of biodiversity compared to other pressing national concerns? How can diverse societies identify these functional roles of biodiversity and assess trade-offs with other public policy objectives?

6.3. Do we know enough about functional roles of biodiversity to be able act?

Biodiversity is the most difficult among our three meso-level environmental issues because there is no clear consensus about the basic functional roles of biodiversity in the landscape. On this, Gowdy (1997, p. 26) points to a dilemma for policy analysts:

> Although our present socioeconomic system cannot continue to expand indefinitely by destroying biodiversity, it is quite possible that economic growth can continue for decades or perhaps even centuries. ... If biodiversity loss and all other forms of environmental degradation will not appreciably affect economic activity in the immediate or even medium-term future, why should we bother to protect it?

The reply also is a question: are some ecosystems headed on a path toward collapse, which, on a human time scale, is essentially irreversible?

The functional role of biodiversity at the local level would appear to be at the beginning of the 'issue life cycle' where there is little awareness of a problem—if indeed one exists—and basic questions about cause–effect relationships have not yet even been identified. How much of what types of biodiversity is needed to maintain productivity and stability? Is it possible to produce a short list of key ecological functions of biodiversity regarding the stability of production systems at the plot or farm level? How about a list encompassing interactions across plots within a landscape? What is the appropriate unit for analysis at the landscape level? What are the appropriate scales—in space and in time—for assessing the effects of biodiversity loss on stability of production systems?

7. Degradation of watershed functions

National concern for natural forest conservation and reforestation often focuses on the degradation of upper watershed functions, typically understood as some combination of:

- on-site declines in *land productivity* as a result of soil erosion,
- off-site concerns about *water supply* (quantity) including annual water yield, peak (storm) flow, dry season base flow, and groundwater recharge or depletion,
- off-site concerns about *water quality*, including siltation of reservoirs and environmental damage from runoff of pesticides, fertilizers, or animal wastes.

7.1. Who cares about erosion?

When is soil erosion a problem for farmers? Can the *on-site impact of erosion on productivity* be measured at the plot level? Can these on-site effects be estimated for bigger units? . . . for landscapes? . . . for states, provinces, or nations? What is an appropriate time scale for such estimates? Are these effects big? If so, under what circumstances? When is soil transfer a problem (or an opportunity) for people downstream? Can the *off-site impact on productivity of soil transfer* (erosion net of sedimentation) be estimated at the landscape level? . . . for states, provinces, or nations?

7.2. Is erosion a policy problem? What about water supply?

Mountain valleys and the great alluvial plains, which are the foundation of food security in Southeast Asia, are products of erosion. What do available estimates tell us about effects of soil transfer on productivity for larger spatial units? Are these values big or small? . . . under what circumstances? On net, does erosion from steep slopes and deposition in the lowlands increase or decrease aggregate production? If erosion were to halt completely, what would be the effect on lowland productivity? How do the net effects on aggregate productivity compare with other effects of soil transfer, siltation of reservoirs for example?

Although 'most analyses of watershed services have focused on soil erosion effects' (Kramer et al., 1998, p. 2), rapid growth in water demand forecast for domestic and industrial uses may over time emerge as a greater threat to growth in food production (Pinstrup-Andersen and Pandya-Lorch, 1998, p. 6; Rosegrant et al., 1997). For Southeast Asia, Rosegrant et al. (1997, p. 7) predict that '. . . a doubling of domestic water withdrawals and a 290% increase in industrial demand will boost the combined share of these sectors in total water demand from 25% in 1995 to 47% in 2020.'

7.3. Does land use change really harm watershed functions?

The question may seem absurd, since there already has been a lot of action. Hundreds of millions of dollars have been spent over decades on soil conservation and watershed management projects in Southeast Asia. Big government bureaucracies exist in most countries in the region to classify perceived watershed 'problems' and to implement a conventional set of 'solutions', typically involving expensive public works, restrictions on land use, and forced eviction of land users. An empirical base has been built to formally justify these actions, including widely accepted and seldom-questioned 'rules of thumb' regarding minimum area under natural forest, maximum slope for agricultural uses, and the like.

Degradation of watershed functions is the most mature of our three environmental topics. Indeed it shows signs of being fossilized at Stage 7 of the 'issue cycle' by vested interests in the present consensus. It remains to be seen how present approaches, and the supporting mindsets, will adapt to regional trends toward decentralization of decision-making, which *may* lead to greater accountability to upland farmers and other local groups, and a spate of relatively new evidence that questions basic relationships between land use change, soil movement, and water supply (Bruijnzeel, 1990; Chomitz and Kumari, 1996; Calder, 1998; Kramer et al., 1998; Lal, 1998; Lindert, 1998). Do landscapes—land uses and their combinations in different patterns or 'landscape mosaics'—matter for soil transfer? How does the sedimentation arising from various landscapes compare with other sources of sedimentation, road construction for example? Do methods exist to quantify erosion from natural processes, agriculture, and other activities (such as road construction) and to assess the impacts (positive as well as negative) of resulting sedimentation at the landscape, provincial, or national scale? Do landscapes differ significantly in their impact on *water supply* downstream? How does land use change affect total water supply (annual yield)? . . . risk and severity of flooding? . . . risk and severity of water shortages? What is an appropriate time scale for such estimates?

8. Concluding question: What can be done about land use change?

If there are 'big' concerns at various scales, what policies and institutional options really can influence the rate and pattern of land use change? Of course past and ongoing policies already have affected land use in Southeast Asia's uplands. These include policies on issues as diverse as resettlement, national defense, road construction, foreign investment, logging, land tenure, narcotics eradication, and agricultural prices. In many cases, there have been big, unintended effects on land use change from policies that were not directed at upland land use at all. Perhaps a prior question—how to improve or redirect the influence of existing policies?—is more pertinent.

But even if existing policies can be reformed to better balance development and a host of other objectives with environmental concerns, a need for new policies and institutional innovations to address specific environmental problems is likely to remain. Policy interventions with explicit environment goals so far have had (at best) a weak influence on specific environmental issues, such as transboundary smoke problems, biodiversity conservation, and watershed management. Banning burning for land clearing has not worked, at least in Indonesia. And while the extent of officially designated protection areas have expanded in some countries—for example, in Cambodia these have multiplied many times in the past decade—the real issue is the effectiveness of protection. In this sense of real outcomes rather than formal designations, effective conservation areas for the protection of biodiversity have continued to shrink in most (perhaps all) countries in the region. And it is difficult to demonstrate results after years of watershed management projects.

Approaches to upland environmental issues so far mainly are variations on land use planning, which seeks to regulate decisions of millions of people dispersed across the landscape, by which we mean combinations of land uses in different patterns or 'landscape mosaics'. Land use planning risks being limited to nice colors on maps in planners' offices but little impact on the ground if it does not also involve workable 'policy levers' that *really* can influence the rate and pattern of land use change that alters these mosaics.

Acknowledgements

The authors have benefited from discussion of these issues with many colleagues, most notably Neil Byron, Mike Swift, and Sampurno Bruijnzeel. Three referees provided useful suggestions for clarifying and improving the paper. This research is part of the global ASB programme, a partnership among national and international institutions to undertake research on sustainable upland systems in various parts of the tropics. This research in Southeast Asia was supported principally by the Asian Development Bank through Regional Technical Assistance Grant (RETA) 5711 and by the Ford Foundation. Additional support was provided by the Australian Centre for International Agricultural Research, The Netherlands, Norway, the United States Agency for International Development, and other donors.

References

Barbier, E.B., 1995. The economics of forestry and conservation: economic values and policies. Commonw. For. Rev. 74, 26–34.
Brenner, A.J., 1996. Microclimatic modifications in agroforestry. In: Ong, C.K., Huxley, P. (Eds.), Tree–Crop Interactions: A Physiological Approach. CAB International (in association with ICRAF), Wallingford, UK, pp. 159–187.
Bruijnzeel, L.A., 1990. Hydrology of Moist Tropical Forests and Effects of Conversion: A State of Knowledge Review. Humid

Tropics Programme, UNESCO International Hydrological Programme.
Calder, I.R., 1998. Water-Resource and Land-Use Issues. SWIM Paper 3. International Water Management Institute, Colombo, Sri Lanka.
Carter, V.G., Dale, T., 1974. Topsoil and Civilization, Revised edition. University of Oklahoma Press, Norman, OK.
Chomitz, K., Kumari, K., 1996. The domestic benefits of tropical forests: a critical review emphasizing hydrological functions. Policy Research Working Paper 1601. Policy Research Department, The World Bank, Washington, DC.
Coase, R., 1960. The problem of social cost. J. Law Econ. 3, 1–44.
Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.
Crissman, C.C., Antle, J.M., Capalbo, S.M. (Eds.), 1998. Economic, Environmental, and Health Tradeoffs in Agriculture: Pesticides and the Sustainability of Andean Potato Production. Kluwer Academic Publishers and the International Potato Center, Lima, Peru.
Daily, G.C. (Ed.), 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, DC.
Economy and Environment Program for SE Asia (EEPSEA), 1993–1998. The Economy and Environment Program for Southeast Asia, Singapore.
Ehrenfeld, D., 1988. Why put a value on biodiversity? In: Wilson, E.O., Peter, F.M. (Eds.), Biodiversity. National Academy Press, Washington, DC, pp. 212–216.
Gowdy, J.M., 1997. The value of biodiversity: markets, society, and ecosystems. Land Econ. 73, 25–41.
Izac, A.-M.N., 1997. Ecological economics of investing in natural capital in Africa. In: Buresh, R.J., Sanchez, P.A., Calhoun, F. (Eds.), Replenishing Soil Fertility in Africa, vol. 51. SSSA Special Publication, pp. 237–251.
Jepma, C.J., Munasinghe, M., 1998. Climate Change Policy: Facts, Issues, and Analyses. Cambridge University Press, Cambridge, UK.
Kramer, R.A., Pattanayak, S., Priyanto, A., 1998. Watershed protection benefits and the Ruteng nature recreation area in Flores, Indonesia. Nicholas School of the Environment, Duke University, Durham, NC.
Krishnaswamy, A., Hanson, A. (Eds.), 1999. Our Forests, Our Future. Summary Report of the World Commission on Forests and Sustainable Development. Winnipeg, Canada.
Lal, R., 1998. Agronomic consequences of soil erosion. In: Penning de Vries, F.W.T., Agus, F., Kerr, J. (Eds.), Soil Erosion at Multiple Scales. CAB International, Wallingford, UK.
Lalley, T., Magnino, L., 1999. Blueprint to solve global forest crisis presented by former top world leaders: world commission proposes coordinated management system. Press Release. Environmental Media Services and Fenton Communications.
Lindert, P., 1998. Shifting Ground: The Changing Agricultural Soils of China and Indonesia. University of California, Davis, CA.
Makarim, N., 1999. Interview by Thomas P. Tomich on 16 April 1999. In: Tomich T.P., Thomas, D.E., van Noordwijk, M. (Eds.), Proceedings of the Methodology Workshop on Environmental Services and Land Use Change, Bridging the Gap between Policy and Research, ICRAF SE Asia and ASB-Indonesia Consortium, Abstracts and Key Policy Issues, Chiang Mai, Thailand, Bogor.
Menz, K., Conroy, C., Gunawan, A., Ellis, K., Grist, P., Bagnall-Oaklely, H., 1997. Fire as an economic disincentive to smallholder rubber planting in *Imperata* areas of Indonesia. *Imperata* Project Paper 1997/6. Centre for Resource and Environmental Studies, Australian National University, Canberra.
Norton, B., 1988. Commodity, amenity, and morality: the limits of quantification in valuing biodiversity. In: Wilson, E.O., Peter, F.M. (Eds.), Biodiversity. National Academy Press, Washington, DC, pp. 200–211.
Pimentel, D., Wightman, A., 1999. Economic and environmental benefits of agroforestry in food and fuelwood production. In: Buck, L.E., Lassoie, J.P., Fernandes, E.C.M. (Eds.), Agroforestry in Sustainable Agricultural Systems. CRC Press, Boca Raton, pp. 295–317.
Pinstrup-Andersen, P., Pandya-Lorch, R., 1998. Recent developments and emerging issues in world food security. Choices (third quarter), 4–12.
Potter, L., Lee, J., 1998. Oil palm in Indonesia: its role in forest conversion and the fires of 1997/1998. Report to the WWF, Indonesia Programme, Jakarta.
Randall, A., 1988. What mainstream economists have to say about the value of biodiversity. In: Wilson, E.O., Peter, F.M. (Eds.), Biodiversity. National Academy Press, Washington, DC, pp. 217–223.
Rola, A., Pingali, P., 1993. Pesticide use, rice productivity, and health impacts in the Philippines. In: Faeth, P. (Ed.), Agricultural Policy and Sustainability: Cases Studies from India, Chile, Philippines, and the United States. World Resources Institute, Washington, DC, pp. 47–62.
Rosegrant, M., Ringler, C., Gerpacio, R., 1997. Water and land resources and global food supply. In: Paper Prepared for the International Food Policy Research Institute's 23rd International Conference of Agricultural Economists on Food Security, Diversification, and Resource Management: Refocusing the Role of Agriculture, Sacramento, CA.
Roulet, M., et al., 1998. Minimata disease in the Amazon: the tip of the iceberg. Sci. Total Environ. 223, 1–24.
Tomich, T.P., van Noordwijk, M., Budidarsono, S., Gillision, A., Kusumanto, T., Murdiyarso, D., Stolle, F., Fagi, A.M. (Eds.), 1998a. Alternatives to Slash-and-Burn in Indonesia: Summary Report and Synthesis of Phase II. ASB-Indonesia Report No. 8. ASB-Indonesia and ICRAF Southeast Asia, Bogor, Indonesia.
Tomich, T.P., Fagi, A.M., de Foresta, H., Michon, G., Murdiyarso, D., Stolle, F., van Noordwijk, M., 1998b. Indonesia's fires: smoke as a problem, smoke as a symptom. Agrofor. Today 10, 4–7.
Tomich, T.P., Thomas, D.E., van Noordwijk, M. (Eds.), 1999. Proceedings of the Methodology Workshop on Environmental Services and Land Use Change: Bridging the Gap between Policy and Research in Southeast Asia, Research Abstracts and

Key Policy Questions, Chiang Mai, Thailand, 31 May–2 June 1999, ASB-Indonesia Report Number 11. Bogor, Indonesia.

Tomich, T.P., van Noordwijk, M., Budidarsono, S., Gillision, A., Kusumanto, T., Murdiyarso, D., Stolle, F., Fagi, A.M., 2001. Agricultural intensification, deforestation, and the environment. In: Lee, D.R., Barrett, C.B. (Eds.), Tradeoffs or Synergies? Agricultural Intensification, Economic Development, and the Environment. CAP International, Wallingford, UK.

United Nations Development Programme (UNDP), United Nations Environment Programme (UNEP), and the World Bank. 1994. Global Environmental Facility: Independent Evaluation of the Pilot Phase. World Bank, Washington, DC.

Vayda, A.P., 1998. Finding causes of the 1997–1998 Indonesian forest fires: problems and possibilities. Report on Preliminary Research. WWF, Indonesia Programme, Jakarta.

Winsemius, P., 1986. Gast in eigen huis, beschouwingen over milieumanagement (‘Guest in one’s own house, thoughts on environmental management’, in Dutch). Samson H.D. Tjeenk Willink, Alphen aan den Rijn, The Netherlands, pp. 227.

World Bank, 1997. World Development Report 1997. Oxford University Press, New York.

World Resources Institute (WRI), United Nations Environment Programme (UNEP), United Nations Development Programme (UNDP), World Bank. 1998. World Resource 1998–1999: A Guide to the Global Environment, Environmental Change and Human Health. Oxford University Press, Oxford.

Zilberman, D., Marra, M., 1993. Agricultural extenalities. In: Carlson, G.A., Zilberman, D., Miranowski, J.A. (Eds.), Agricultural and Environmental Resource Economics. Oxford University Press, Oxford, UK, pp. 221–267.

Available online at www.sciencedirect.com

SCIENCE DIRECT®

Agriculture, Ecosystems and Environment 104 (2004) 19–34

Agriculture Ecosystems & Environment

www.elsevier.com/locate/agee

Quantifying off-site effects of land use change: filters, flows and fallacies

Meine van Noordwijk[a,*], John G. Poulsen[b,1], Polly J. Ericksen[c,2]

[a] *World Agroforestry Centre (ICRAF), SE Asia Regional Program, P.O. Box 161, Bogor 16001, Indonesia*
[b] *Centre for International Forestry Research (CIFOR), Bogor, Indonesia*
[c] *ASB Global Coordination Office, ICRAF Headquarters, Nairobi, Kenya*

Abstract

Many external effects of land use change are based on modifications of lateral flows of soil, water, air, fire or organisms. Lateral flows can be intercepted by filters and thus the severity and spatial range of external effects of land use change is under the influence of filter effects. Wherever lateral flows are involved, research results cannot be simply scaled on an area basis, and overall impact does not follow simple linear causal relationships. This complexity has consequences for relationships amongst the primary agents who initiate or exacerbate external effects, other stakeholders who are affected by them and policymakers who attempt to mitigate problems that reach sufficient visibility in society. In this paper we review how the relative importance of lateral flows and filter effects differs among a number of externalities, and the implications this has for research methods. If flows and filters are incompletely understood, policies may be based on fallacies. Whereas 'fire-breaks' act as filters in the lateral flow of the high temperature pulse of a fire, smoke from land-based fires can be intercepted only by rainfall acting as a filter and the external impact of smoke is determined by the atmospheric conditions governing lateral flow and chemical transformations along the pathway. Causal relations in smoke and haze problems are relatively simple and may form a basis for designing policy interventions to reduce downwind damage. For biodiversity issues, landscape connectivity, the absence of filters restricting dispersal and movement of organisms, is increasingly recognised as an influence on the dynamics of species richness and its scaling relations. Biodiversity research methods can extend beyond the current descriptive stage into clarifying causal relations with a lateral flow perspective. The question whether connectivity is in fact desired, however, depends on stakeholder interests and situation. Forest functions in watershed protection, presumably leading to a continuous flow of clean water in the dry season through the subsoil instead of a rapid surface transfer, have been generally attributed to the trees rather than the forest, with its rough surface structure, swamps and infiltration sites. A new synthesis of site-specific hydrological knowledge and tree water balance studies may be needed to separate myth from reality, and avoid wasting public funds on tree planting under the heading of reforestation, without restoring the hydrological regime of a real forest. Soil movement can be intercepted at a range of scales and in as far as soil transport entails movement of soil fertility, filter zones can be very productive elements of a landscape. To achieve 'integrated natural resource management' all external effects of land use will somehow have to be taken into account in farmer decision making about the use of natural resources on and off farm. Farmers' ecological knowledge may include concepts of lateral flows and should be further explored as an integral part of a new landscape ecological approach.

Keywords: Biodiversity; Filters; Fire; Lateral flows; Scale effects; Watershed functions

* Corresponding author. Tel.: +62-251-315-234; fax: +62-251-315-567.
E-mail addresses: m.van-noordwijk@cgiar.org (M. van Noordwijk), ericksen@iri.columbia.edu (P.J. Ericksen).
[1] Present address: Wagnersvej 9, 7400 Herning, Denmark.
[2] Present address: International Research Institute for Climate Prediction, Columbia University, New York, USA.

0167-8809/$ – see front matter
doi:10.1016/j.agee.2004.01.004

1. Introduction

The questions raised by Tomich et al. (this volume) imply a need for methods for quantifying effects of land use change across a hierarchy of scales. The economic concept of 'externalities' relates to effects outside of the analysis by the decision maker. Often, but not necessarily, these are effects at some physical distance, external to the land unit directly affected by the decision, depending on the scale and organisation of human land use (Sinclair, 1999a). Many externalities are based on (changes in) lateral interactions between land units. Lateral interactions may consist of mass flows of soil, water or air, of specific substances and organisms carried in such flows, or of active movement of organisms. Wherever 'lateral interactions' play a role, area is not an unequivocal basis for expressing results of measurements and scaling is not a trivial exercise of multiplying total area by average value per unit area. Externalities based on lateral interactions can have a complex causal relationship, as effects can be mitigated or influenced by filter functions of landscape elements at intermediate scale (Fig. 1). The location of filters is likely to be at least as important as the total area available to the land cover types that can exert this function. If flows and filters are incompletely understood, however, policies may be based on fallacies. Understanding the filters and flows, however, is a necessary but not sufficient condition for effective governance.

Trees and patches of forests can play such a filter role, giving a new dimension to research in agroforestry (Van Noordwijk and Ong, 1996). Filters can discouple flows of dissolved particles from a mass flow of water, but also act on flows of air or even organisms. Approaches of the landscape mosaic from above (remote sensing, patterns, land use planning) are complementary to those from below (lateral interactions, transport processes, farmer management decisions, stakeholder–policy–agent feedback). A landscape is here defined as a heterogeneous area made up of a cluster of interacting ecosystems/habitats, usually repeated in similar form in a regional context (Forman, 1995). Landscape structure is defined as the sizes and shapes of these patches of different habitat types (here used interchangeably with ecosystems), and the distances of these patches from one another. In agro-ecosystems the landscape organisation is closely linked to the patterns of human organisation. We will review how spatial patterns in a landscape mosaic affect the processes of lateral interactions which these externalities entail and hence the degree to which intermediate or landscape level solutions can reduce the conflicts of interest between private land use decisions and values regarded by society at large and/or specific interest groups of external stakeholders.

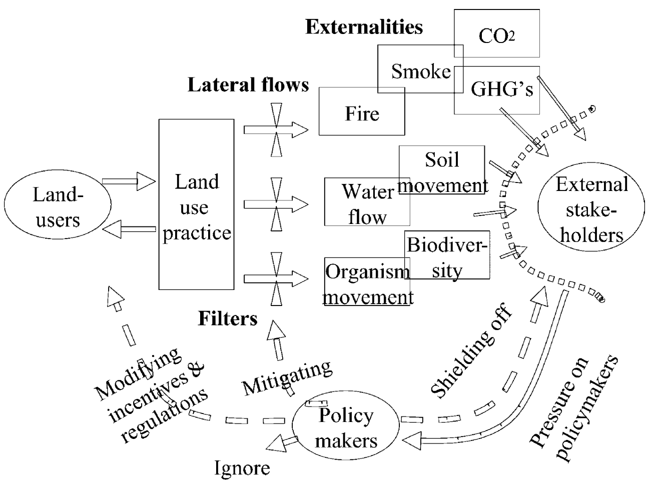

Fig. 1. Schematic relationships between land use practices, lateral flows, filter effects and external impacts, and the feedback loop from stakeholders via policymakers to efforts to modify the land use, strengthen the filter functions or shield off the external stakeholders; GHG's: greenhouse gasses.

Table 1
‘Externalities’ or land use effects beyond farm level, classified by categories of lateral movement

What moves?	Examples	Approximate range (km)	Causing what type of (+ or −) externality?	Can flow be stopped? (++ = easily, −− = not at all)	How?
Soil/earth	Landslides	0.1–1	Physical destruction (−)	+	Forested strips as filter
	Water-borne sediment	10–100	Siltation of reservoirs (−), fertilisation (+)	++	Riparian strips and vegetative filters
	Air-borne dust	100–1000	Fertilisation (+/−)	+	Windbreaks
Water/solutes	Floods	10–100	Drowning and destruction (−)	+/−	Riparian zones and floodplains
	Dry season river flow	10–100	Off-season water supply (+)	−/+	Reservoirs
	Total river water yield	10–1000	Water supply (storable) (+)	−/+	Groundwater use
	Groundwater recharge	10–1000	Off-site water supply (+)	−/+	Landscape surface roughness and infiltration sites
	Salt	1–10	Salinisation (−)	−/+	Salt absorbing vegetation
	Nutrients	1–100	Eutrophication (−/+)	+/−	Absorptive filter (‘safetynet’)
	Pesticides and other chemicals	10–1000	Pollution (−)	−/+	Absorptive (biological) filter
Air	Wind	0.1	Abrasion (−)	++	Windbreaks
	Greenhouse gases	Global	Greenhouse gas effect (−)	−−	
	Sulphurous and nitrous oxides	1000	Acid rain (−)	−	
	Smoke	1–1000	Smog, low visibility (−)	−	
	Air humidity	0.01–0.1	Less evapotranspiration (+)	+	High evaporation strips
	Water-vapour	10	Effects on rainfall? (−)	−/+	
Fire	High temperature pulse	1–10	Destruction and burn (−)	+	Fire-break
Organisms	Free roaming predators	0.1–10	Reducing pest outbreaks (+)	−/+	Lack of corridors connecting to refugia
	Pollinators	0.1–1	Securing fruit set (+)	+	Lack of nearby patches with host plants
	Desirable (forest) species	0.1–1	Providing spontaneously established resources (+)	+	Lack of connections to nearby refugia
	Pests and diseases	0.1–1	Yield loss/crop failure (−)	+/−	Filters = interrupted corridor
	Weeds	0.1–1	Yield loss/crop failure (−)	+/−	Filters = interrupted corridors
	Soil ‘engineers’	0.01–0.1	Repairing soil structure (+)	+	Lack of nearby refugia for recolonisation

The main categories of problems discussed by Tomich et al. (this volume), watershed functions, smoke and biodiversity can be classified by different phases of transport: movement of earth, water, wind, fire or organisms (Table 1). Spatial relations caused by the flow of water, air and moving organisms were discussed for NW Europe by Vos and Opdam (1993).

The data required to have impact on human decisions and policies, and hence the most appropriate methods to obtain relevant data, depend on where the given 'externality' is on the 'issue life cycle' (Tomich et al., this volume).

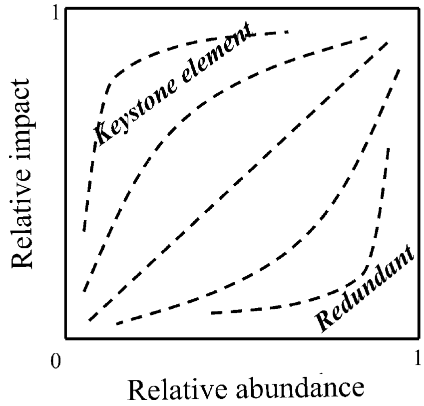

Fig. 2. Definition of 'keystone' elements of a landscape based on their large impact relative to the fraction of area occupied.

2. Externalities, environmental service functions and filters

Land use change can impact on the service functions (Constanza et al., 1997) such as the supply of clean air and water, which everybody expects the environment to provide, but few want to take effort to maintain. Land use systems can be classified by their influence on on-site as well as off-site environmental service functions (Table 2).

The term 'filter' is here used in a generic sense of anything that can intervene with a lateral flow. Typically, filters occupy a relatively small fraction of the total area and have a large impact per unit area occupied. They can thus be regarded as 'keystone' elements of a landscape (Fig. 2). Filter elements can be easily missed out in remote sensing approaches, but should be the focus of research if we want to understand how the landscape functions as a whole.

Closely coupled to the issue of filters and flows is the question of whether spatial pattern matters. A comparison of landscapes with the same relative area occupied by trees but in different spatial configurations has been at the core of 'agroforestry' research, establishing where agroforestry can be superior to the weighted mean of its 'agriculture' and its 'forestry' component (Sinclair, 1999b).

Table 2
Environmental service functions of landscape elements at a range of scales (modified from Izac and Sanchez (1999))

	Farm	Landscape	Region	Global
Harvested net primary production (NPP)	Food production and income generation	Food and fuel security, poverty alleviation	Food and fuel security, poverty alleviation	Reduce mass poverty
Non-harvested NPP	Soil resource conservation	Soil and forest functions	Forest functions	Carbon storage
Nutrient use and replenishment	Nutrient cycling, decomposition and mineralisation	Nutrient cycling, lateral flows in mosaic	Preventing depletion and excess (pollution)	
Soil movement	Erosion control and sediment retention	Soil transfers (losses and gains), characteristics of streambeds and lakes	Preventing siltation and pollution	
Water use and replenishment	Infiltration and use of soil water	Streams and subsurface water flows	River flow, aquifers	
Climate regulation	Microclimate for crops and animals	Effects on air turbulence and rainfall distribution	'Teleconnections' via circulation cells	Greenhouse gas concentrations
Facilitating other biota	Pollinators, biological control agents	Pollinators, biological control agents, refugia	Refugia	Biodiversity

3. Methods for scaling lateral flows

Quantification of external impacts of land use can be approached in different ways (Table 3):

1. Approaches based on 'balance sheets' for well-defined land units (such as C-stocks, nutrient balance, water balance, and local species richness).
2. Approaches based on direct measurement of transport (mass flow of carrier plus concentrations, organism dispersal) across land units.
3. Approaches based on 'filter elements' in the landscape known to interact with lateral interactions (this may be a special form of 1).
4. Measurements of intensive parameters at a range of scales and analysis of apparent 'fractal dimensions', as indicators of lateral interactions (see below).

The use of at least two of these categories of methods may be an important consistency check (especially in early stages of the 'issue life cycle'), and may help in evaluating cost-effective monitoring schemes for routine applications.

In many disciplines past approaches were based on the notion of a 'representative elementary volume' (soil science), 'minimum sample area' (biodiversity) or 'representative farm households within agro-ecological zones' (social science and farming systems), as a unit which contained the salient properties of the system to be studied. Within such a unit, one typically assumes complete mixing, while between units substantially less exchange would occur. Overall properties are calculated by multiplying total area (or volume) with the established average value per unit area (or volume). Although valuable as a first approximation, no universal delineation of such units can be found that transcends disciplines and properties.

During the last few decades, progress was made to transcend the earlier delineation debates. Over a certain range of scales, relatively simple rules were found to apply for the scale impact. These rules may, within a certain scale range, be of the form:

$$Y_L = Y_1 \left(\frac{L}{L_1}\right)^a, \tag{1}$$

where Y_L and Y_1 are system properties at length scale L and L_1, respectively; and a is the dimension. If a is 1, the scaling rule is linear; for $a = 2$ or 3, the rule is area or volume based; if a is not an integer, the rule is 'fractal'. The fractal dimension of species richness, for example, has been under study for over 30 years in the form of the theory of island biogeography (Rosenzweig, 1995; see below). Fractal approaches have found wide application in geography (Lam and de Cola, 1993) and landscape ecology (Farina, 1998).

Fractal properties ('dimension') apply to 'self similar' systems across scales, and can be used in extrapolating measurements of limited sample points to quantitative statements about system properties at any scale within the range where the rules apply. A fractal dimension can thus serve as a simple summary parameter for the (spatial or temporal) scaling properties of a systems attribute, and knowledge of its magnitude is often at least as important as having a precise estimate of system attributes at a particular scale of measurement (Van Noordwijk, 1999a). Crawford et al. (1999) discuss the contribution of fractal models to the integration of processes in soils, with an emphasis on soil physical properties, but also some first applications to soil biology. Recognition of the appropriate scaling rules may help to understand the risks of 'scaling up' essentially plot level nutrient

Table 3
Generic approaches to measurement of 'intensity' and 'extent' parameters of environmental functions of landscape elements

	'Intensity' parameter (amount per unit area or volume)	'Extent' parameter (total area or volume influenced)
Balance sheet	Amounts per unit land in a category	Land area in different categories
Fluxes and flows	Concentrations of sediment, soot particles or gasses	Mass flow of water or air carrier
Filters	Maximum filter function per unit time per unit filter element, its saturation and subsequent regeneration	Quantity and location of filter elements
Determination of fractal dimensions	Relative contribution of lateral flow in overall process	Empirical relations between properties measured at different scales

Table 4
Contrast between stratified sampling approach and landscape scaling approach to estimate total value of entities over a land unit

Stratified sampling	Landscape scaling
1. Identify internally relatively homogeneous, mutually independent strata	1. Identify 'landscape functional types' as mosaics of interacting elements
2. Estimate the typical value for each stratum (y_i)	2. Estimate the mean value at a certain unit scale (Y_1)
3. Establish the area for each stratum (f_i)	3. Establish the fractal dimension (z) by repeating step 2 at other scales
4. Multiply areas and value to get overall results: $Y_t = Sf_i y_i$	4. Scale plot results to any landscape scale s and associated area A_s: $Y_s = Y_1 A_s^z$
	5. If results are to be added for multiple landscape types, estimate interaction term: $Y_t = SY_{i1} A_{it}^{zi}$

balances to the African continent (Van Noordwijk, 1999b). The essential steps in approaches based on 'stratified sampling' and those based on 'landscape scaling' are summarised in Table 4.

The impact of measurement scale on the outcome of a measurement (even if expressed 'per unit area') may be counter-intuitive: as long as one makes sure that all elements of a population have equal chance of being represented in the measurement, one would expect sample size to influence the confidence interval, but not the mean result. Two examples may illustrate our point. First of all consider measurement of human migration, using a different measurement units, e.g. homestead, village, district, province, nation, continent, planet. Even if one makes sure to include all human beings just once in the sampling, the result expressed as fraction of migrants as part of the total population will strongly decrease with sample size, from close to 100% at home (stead) level to 0% at the planet scale. The increase of sample size has 'internalised' most of the border crossings, which define migration. What applies to human migration, applies similarly to migration of other species and explains part of the complexities in scaling biodiversity measures. Secondly, consider erosion. Plot level erosion may be high under many agricultural practices, but at a continental scale Africa loses hardly any sediment to seas and oceans, as most sediment will be trapped within terrestrial, riverine or lake habitats.

The connection between lateral flows and fractal dimensions can be further explored on the basis of this erosion/sedimentation example, by constructing maps of positive (net erosion sites) and negative (net sedimentation sites) numbers, with for example a random distribution. This map can be sampled at a range of scales (all of them covering each cell in the map just once), with the additional rule that a sample reflects the average value of the cells it contains, but cannot be negative. This latter rule is based on the conventional approach in erosion measurements where incoming sediment flows are excluded from measurement plots, and net sedimentation is thus represented by a zero result. Using such a method, the end result will be that at a measurement scale of a single unit all negative values are perceived as zero and the average result will overestimate the real value. With increasing sample size the average per unit area will decrease, until a sample size is reached at which no individual samples produce a zero result.

A plot of the logarithm of the result against the logarithm of the sample scale may produce a straight line over part of the range (fractal scaling), levelling off at some point (multiple scaling). As illustrated in Fig. 3, the slope of the line and hence the fractal dimension of the process will depend on the frequency of negative results. Scale effects over a larger part of the range are to be expected if non-random patterns of positive and negative values are introduced. A direct analogy may exist between this example and the measurement of erosion and other lateral flow surface phenomena where the filters are not counted as 'negative flows'. There is nothing mysterious about these scaling phenomena, but we are so familiar with the erroneous results of methods ignoring these issues that it takes effort to digest it.

In the remaining part of this article, we will briefly review methodological issues for the three types of externalities considered in this volume.

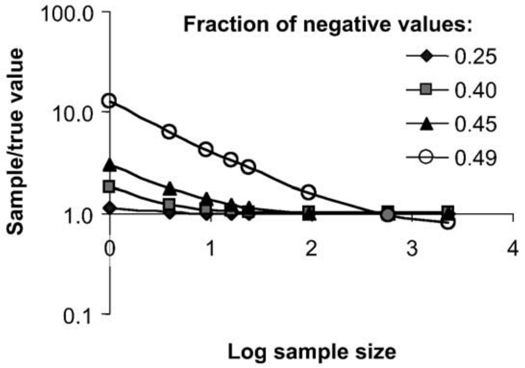

Fig. 3. Effect of measurement scale (log form) on average measurement result (log form) for a random grid (48 × 48 cells) of positive and negative values, depending on the fraction of negative values (results averaged over 10 replicate randomisation's); for further explanation see text.

4. Quantifying external effects of smoke and fire

Transboundary haze problems as experienced in SE Asia appear to be the direct result of biomass burning (forest fires, land clearing by slash-and-burn, residue disposal, burning and smouldering of peat soils and superficial coal deposits). The debate is focussed on the reasons why people set fire to particular pieces of land (‘fire as a tool, or as weapon’, Tomich et al., 1998), not on the relation between fire and smoke per se. As impacts of haze are rapid and source areas can be traced by remote sensing (Murdiyarso et al., this volume), it seems that the conditions are right for a direct feedback from the stakeholders affected by the haze and smog to the land users responsible, with national policymakers involved where international relations are involved. On closer inspection, however, even this issue has lateral flow and filter aspects that may complicate policy formulation and implementation.

Malingreau and Zhuang (1999) reviewed the global significance of biomass burning. Non-methane hydrocarbons emitted by vegetation apparently play a role in the chemical transformations (including ozone formation) in the lower atmosphere, converting the primary emissions of methane, carbon monoxide and nitrous oxides to the forms, which have impact elsewhere. Rainfall is the major filter, discoupling airflow from the gasses and particulate matter it carries along. The 1997/1998 smoke and haze impacts were aggravated by a lack of rainfall, an inversion situation where the normal vertical escape of air flows to higher atmospheric layers did not occur and admixture of urban and industrial emissions to the smoke and primary fire products. The relation between point source of smoke and other products of biomass burning and impacts elsewhere may not be as straightforward as it appears at first sight, but still be sufficient for policies to focus on the primary agent.

For the lateral flow of an energy pulse leading to the spread of fire, the filter function of a fire-break is a well-recognised part of landscape level design rules. Any spatially explicit model of fire occurrence needs rules for fire initiation and extinction in each landscape unit (e.g. ‘pixel’) and the probability of fire spread from neighbouring units. A basic form for the probability of fire occurrence in any part of a landscape could be (compare Wibowo et al., 1997):

$$p(\text{fire})_i \quad \text{(fire initiation in a land unit)}$$
$$= (I - E)FW + \sum_{j}^{8} p(\text{fire})_j (1 - F_{\mathrm{A}})(1 - F_{\mathrm{B}})$$
$$\text{(spread from neighbourhood)} \qquad (2)$$

where I is the p(initiate), E the p(extinguish), F the relative fuel factor (0–1), W the relative weather factor (0–1), F_{A} the abiotic filter (0–1) and F_{B} the biotic filter (0–1).

The terms I and E primarily depend on the social actors involved and the incentives they face in initiating or extinguishing fires (in general, or under specific conditions). Research methods to quantify the biophysical model terms (F, W, F_{A} and F_{B}) are available and can be coupled to remote sensing characterisation of the connectivity of ‘burnable’ pixels. The longer-term impacts of fire on ecosystems depends strongly on the ecosystem and stakeholder perspective (Whelan, 1995).

5. Quantifying effects of land use change on biodiversity

As evident of the mix of + and − signs in Table 1, movements of organisms across a landscape can be deemed desirable or can pose a threat. Any change, either increasing or decreasing connectivity can change the biotic component of any location, leading to ‘externalities’. In terms of the ‘issue cycle’ (Tomich et al., this volume), we see on one hand a strong drive for policy interventions (e.g. a global convention on biological diversity), yet a large degree of uncertainty and confusion still exists on causal relations and what exactly we want to protect for what reason. That discrepancy makes any implementation of where and how to apply such protection policies hard to defend where it conflicts with other interests.

No single ‘correct’ scale exists for describing populations or ecosystems (Bunnell and Huggard, 1999). In the lateral flow terminology, the organisms themselves move whenever the landscape provides the right type of continuity (by air, land or water for various categories of organisms). A range of measurement and modelling approaches exists for population redistribution in animals and plants (Turchin, 1998). The positive or negative interpretation of such lateral flows largely depends on the local or exotic nature of the organisms. Global biodiversity is largely due to geographic isolation and reducing such isolation can be clearly undesirable, even at a continental scale. On the other hand, fragmentation of previously continuous forests forms a clear threat for the survival of meta-populations (Harvey and Haber, 1998). Maintaining or re-establishing an ‘ecological infrastructure’ in derived landscapes has become a main issue in the developed, temperate parts of the world (Opdam et al., 1993). Applications in the tropics are relatively scarce. Harvey (2000) reported how ‘windbreaks’ can provide dispersal pathways for trees via seed-eating birds, but that small (e.g. 20 m) discontinuities among forests and windbreaks can have a large impact.

Until recently (Hubbell, 2001), the most coherent theoretical framework for biological diversity (Rosenzweig, 1999) was formed by the theory of island biogeography, relating species richness S at any scale s to the richness at unit scale S_1 and a fractal dimension z: $S_s = S_1 A_s^z$. This approach first of all applies within a single type of landscape. Where multiple scaling is involved, as in the consideration of regions with multiple landscape types, a summation over the different landscape types should include a correction for the degree of overlap in species composition. The parameters in this equation have been interpreted from the balance of local extinction and recolonisation, depending on the connectivity with other suitable habitats for the species group considered (Rosenzweig, 1995). Alternative interpretations are currently debated (Harte et al., 1999; Rosenzweig, 1999; Hubbell, 2001). Species–area curves obtained in sampling within connected landscapes, however, cannot be used to predict the impacts on species richness if areas are modified, as species richness for areas which have become islands will decline over time once recolonisation rates are reduced—this means that the fractal dimension z is time (or context) dependent (Kramer et al., 1997; Kunin, 1998; Rosenzweig, 1999).

Hubbell (2001) made a valiant attempt at unifying the theory of biodiversity and that on biogeography. His theory is ‘neutral’, in the sense that it does not rely of any differences in attributes or functions of the species that inhabit his model world—except for being identifiable by a different name or code. Yet, the theory gives an efficient (based on only a few parameters) account for many datasets on species richness of tropical forest trees, birds and insects across widely different scales. The theory builds on to the framework of the theory of island biogeography (dispersal limitations, population size effects), but ‘unites’ this with ideas that the current species richness of the world is a balance between the rates of ‘speciation’ and ‘extinction’. As Hubbell states, there are essentially two views on why diversity exists:

- A ‘niche assembly’ theory that assumes that species can only survive competition by being ‘sufficiently different’ from the others and ‘occupying an at least partly different niche’ (i.e. intraspecific competition is supposedly stronger than interspecific competition).
- A ‘random walk’ or ‘transient’ theory that states that the numbers of all existing species tend to go up or down, and that the total diversity in any given space and time sampling frame is just a matter of chance—with, however, reasonably tight predictions about the relative frequencies of differently ranked species at different scales. The random walks of all species can, however, be constrained to maintain a constant total density of individual organisms (e.g. constant tree density per unit area), reflecting overall resource availability constraints (and hence the impacts of both intra- and interspecific competition).

Hubbell (2001) claims that the latter framework may be sufficient to account for (nearly) all that we know quantitatively. Of course this does not disprove the 'niche assembly' concept, it only makes the concept 'redundant'. The new framework has consequences for sampling, data collection and data analysis—but can it also inform the human value problem of our efforts to slow down the current rate of biodiversity loss? The 'predictions' of Hubbell's theory on conditions for maintaining species richness are a combination of 'meta-populations', dispersal rates and connectivity, that have essentially been appreciated over the last few decades—so it does not bring many surprises from that perspective (it does add some precision to the predictions of time-patterns of losses after fragmentation). More importantly, probably, is that the theory may 'undercut' the popular functional interpretation of diversity. When current or potential future human use of specific properties of organisms are involved, the reasons for maintaining diversity of course remain valid. Where we say, however, that diversity is essential for the normal functioning of ecosystems, we may need to more carefully phrase what is meant by 'functionality'.

Noble (1999) discussed filters and concentrators in the context of landscape fragmentation and mobility of species. Animals appear, grosso modo, to be more sensitive to fragmentation and human disturbance than plants, but secondary impacts on plants may occur via animal partners required for pollination or seed dispersal, or via modified herbivore impacts on vegetation succession.

While loss of any species or genetically distinct population by definition depletes the genetic library (and hence its potential for supplying direct economic benefits to society), each extinction also has the potential for generating cascades of further losses. Although ecosystem services on a global scale are expected to depend on population diversity (Ehrlich and Daily, 1993; Daily and Ehrlich, 1994), the connection between diversity of populations and the delivery of ecosystem services at local and regional scale remains yet to be clarified. The ability of a monoculture to maintain services over a long time is subject to debate (Anderson, 1994; Vandermeer et al., 1998). Although monocultures (especially those that maintain genetic diversity in space, or maintain a rapid turnover in their genetic make-up by frequently replacing the germplasm used) may provide many ecosystem services over decades/centuries, they may be more vulnerable to catastrophic disease and/or be less resilient in the face of environmental change. Furthermore, the drastic reductions in species diversity in an ecosystem may lead to sequences of community development whose direction and consequences for ecosystem services may be very difficult to predict (Drake et al., 1993). Yet, evidence that species richness contributes directly to ecosystem maintenance and function at large is scant and inconclusive (Simberloff, 1999; Vandermeer et al., 1998). Biodiversity conservation is defensible as an end in itself; its more local role as a means to 'forest health' or agro-ecosystems resilience in an immediate neighbourhood has not yet been established (Simberloff, 1999; Kramer et al., 1997).

Considerable overlap exists between the kinds of species most sensitive to spatial structure (top predators, other large area-sensitive species, late-successional species), and those species most likely to have large influences on their ecosystems. This overlap suggests that changes in spatial structure can potentially have serious consequences at the level of ecosystem organisation.

To analyse the functional significance of biodiversity we may need to tease apart:

1. the biological and ecological organisation of a landscape and their interactions, the number of different biological units at each level of organisation;
2. the degree of similarity (overlap) in the traits or roles that biological and ecological units can play within each organisational level;
3. the spatial configuration and its influence on individuals (foraging, dispersal) and meta-populations (local extinction and recolonisation; Harrison, 1994).

A strict 'externality' version of the biodiversity conservation question is: what is the impact of biodiversity at location A, on the functioning of an agro-ecosystem at B. Is it important for B to maintain a forest at A, provided that the essential parts of the genetic library of A remain accessible elsewhere? Many of the presumed ecological neighbourhood functions become less obvious the bigger the difference in biota between the forest and the agro-ecosystem. Many forest species, and especially the forest specialists considered most valuable from a conservation perspective, have little role to play in an agro-ecosystem. General predators (cats and snakes) may help to keep rats and mice populations in a rice field surrounded by forest at an acceptable level, but wild pigs, monkeys and elephants coming out of that same forest will make up for the difference. The presence of big cats (e.g. tigers) does not lead to a friendly perception of a neighbouring forest by local farmers, even if they acknowledge that tigers play a positive role in pest control.

6. Quantifying external effects of land use via soil and water flows

In view of the issue cycle (Tomich et al., this volume) soil and water conservation may seem to be an issue that has been largely resolved and at the end of its cycle. Many policies and regulations exist, and substantial incentive structures have been created to modify farmer's choices in land use. One may go so far as stating that considerable vested interests have been established in certain forms of soil conservation and watershed reforestation projects. Yet, the issue seems to be at the start of a new cycle, as the presumed causal relationships on which many current polices are built do not live up to scrutiny. A new wave of research efforts has started, with pioneers such as Hamilton and King (1983), Bruijnzeel (1990, 1997) and Diemont et al. (1991), and researchers attempting to tell the public at large and the policy community that their mental models may be myths (Calder, 1998).

For the public debate on water resources in SE Asia simple questions appear to remain unanswered:

- Are lowland (capital) cities frequently flooded because the uplands are deforested, or because they are situated on 'floodplains' at the mouth of the main rivers?
- Is the recharge of subsurface water flows and off-season streams by 'old-growth forests' due to the *trees* or to the *forest* with its surface roughness, swamps, and lack of channels? What does this mean for 'reforestation' instead of 'planting trees'?
- Is the water use of 'fast growing' trees (such as Eucalypts) more than proportionate to their growth rate (in the absence of C4 trees), and from where do they obtain their water (Calder, 1998)?
- What is the most effective location for 'watershed protection' forests: top-down (covering the hilltops) or bottom-up (primarily aligning rivers and streams; Van Noordwijk et al., 1998)?
- Is the relatively high sediment delivery to marine systems in SE Asia simply due to the relatively short rivers in a geologically young landscape, or does it indicate a strong human impact?.

All SE Asian watersheds are 'exorheic' draining to oceans, as opposed to the 'endorheic' ones draining to lakes and fans, common in more continental parts of the world (compare Mungai et al., this volume). The SE Asian islands alone contribute about 20% of the world's sediments to the marine system (Hu et al., 1999). Lakes and reservoirs are the ultimate filter, discoupling the flow of sediment from the flow of water by reducing the velocity of flow to allow sedimentation. Global net sediment delivery to marine systems may have increased under human influence, but dams in many of the major rivers have led to dramatic local decreases, with all its consequences for fisheries and coastal geomorphology. Meybeck et al. (2001) concluded that more than 25% of global sediment flows already are trapped in reservoirs, but land ocean transfers of N & P increase (while Si decreases, with impact on marine diatoms). Both increases and decreases of land–ocean transfers may have negative impacts on current marine systems. Irrigation engineers aim at reducing all river flows into the ocean to virtually zero (like the situation in the Nile and Colorado river) and using all freshwater for irrigation. At a continental scale terrestrial evapotranspiration is thus increasing, and this may reduce or reverse any effect deforestation may have on rainfall by reducing local evapotranspiration.

Forests can generate subsurface flows of water, and conventional techniques for measuring incoming and outgoing water flows at the soil surface can quantify amount, but many studies so far ignore *incoming* subsurface flows (Wenzel et al., 1998). Downslope lateral flow of water, either over the surface or below, is a major determinant of the coherence of landscapes. Existing models do a poor job on the subsurface part (Wood, 1999), unless specifically parameterised for a given region, as details of the pathway and variations in hydraulic conductivity matter. Human impact on these subsurface flows is little understood in general, but the position of deep-rooted trees in the landscape can often have significant influences on total water flow and the pathway of flows. In landscapes with subsurface salt deposits (such as SW and SE Australia, but also NE Thailand), the pathway itself is a major issue. Lateral flows at the surface, and the sediment they carry, are easily observed and get much attention. Subsurface flows are not visible and are associated with often substantial buffers, causing delays in the cause–effect chain and making it unlikely that policies to modify such flows will have an appreciable and appreciated impact at a politically significant time scale. Yet, groundwater flows can be ecological time bombs that cannot be easily controlled if issues are not resolved at the source. Layers of parent material and soil horizons of different textures have a profound influence on subsurface water flow. Soil texture and parent material influence the flow of water, as do landform shape and location (Gerrard, 1990). Landform shape, e.g. convex or concave, determines how surface water flow is channelled over both the landform in question and the surrounding landforms (Ericksen and McSweeney, 1999). Breaks in landforms, or the location of a concave footslope below a convex backslope can serve as filters, and have been exploited for agriculture, the world over. Landscape level hydrological models are needed for the details, but equally important are simple ways to 'read the landscape' for the extrapolation phase of policy action.

Forest functions in watershed protection, presumably leading to a continuous flow of clean water in the dry season, have been generally attributed to the trees in stead of the forest, with its rough surface structure, swamps and infiltration sites. A new synthesis of site-specific hydrological knowledge and tree water balance studies may be needed to separate myths from realities, and avoid wasting public funds on tree planting under the heading of reforestation.

Soil movement can be intercepted at a range of scales and in as far as soil transport entails movement of soil fertility, filter zones can be very productive elements of a landscape, at least partly offsetting loss of productivity in erosion zones (Daniels and Nelson, 1987). Little is known of the regeneration capacity of biological filters in riparian strips after temporary saturation (Lowrance, 1998).

Erosion/sedimentation research has to expand from its traditional focus on small plots (Stocking, 1998; Evans, 1993; Watson and Evans, 1991). The huge variability of soils at landscape scale, however, forms a major challenge in separating land use impact from existing background variation. Kabrick et al. (1997), restricted their samples to particular landforms, to eliminate this source of variability, and then used directional transects with logarithmically spaced intervals to sample soils, ultimately relying upon spatial variograms to estimate the correlation of soil properties with distance. Variograms are the basis for kriging, which is the tool used for the spatial extrapolation of soil properties (Burrough, 1993) to predict the characteristics of unsampled areas, along with multiple linear regressions and other more standard tools. Moore et al. (1993a) successfully predicted 70% of the distribution in soil attributes from variation in terrain attributes. Pennock et al. (1994) quantitatively assessed the impact of cultivation on soil quality over a landscape using a digital elevation model (DEM) to develop landform classification units. Moore et al. (1993b) discuss the methods available for predicting water flow and sediment redistribution. These methods have been used in a number of recent studies, e.g. Grunwald and Frede (1998). The limitation to these models are the significant data requirements, and the difficulties of using them in highly irregular environments.

Landslides are triggered (Iida, 1999) when the weight of the saturated soil column exceeds a critical value of friction on a plane of weakness, modified by degree of anchoring provided by deep tree roots. As landslides are essentially caused by subsurface flows, their frequency may depend less on land cover than commonly believed. In closed forests, however, landslides may have less downstream impact than in agriculturally used landscapes because forests may provide more effective filters around the streams. Road building has an obvious direct impact on landslide frequency, probably exceeding its indirect impacts via associated land use change (Ziegler et al., this volume). The strong connectivity provided by roads allows for high sediments delivery rates to streams of erosion products associated with roads, unless technical designs provide adequate filters.

Many studies have now documented that sediment flows in rivers are not as closely linked to ongoing erosion in uplands as previously thought. Modelling tools that include both agricultural and non-agricultural sources of sediment are now more widely in use (Baffaut et al., 1998). Careful landscape reconstructions can lead to a reconsideration of 'blame' attributions, and more importantly lead to more effective interventions to protect downstream interests. For example, Fryirs and Brierley (1999) described how the major land use change caused by European settlement in SE Australia led to the development of continuous channels in previously discontinuous river courses, greatly increasing sediment delivery from the catchment. Although disturbance of slopes resulted in significant movement of soil, most of this material was stored on-slope, in trapped tributary fills and along lower order drainage lines, as the slopes were effectively decoupled from the river channels. The sediment flowing out of the catchment largely originated from the riverbeds rather than from current erosion on the slopes. Tongway (1990, 1994) explored the role of vegetation in the degradation and restoration of rangelands, via the trapping and channelling of water and sediment. These patches range from clusters of lichens to shrubs and trees. Risser (1989) quantified the different capacities of vegetation units within a landscape to trap nutrients, patches and water, as a function of morphology, rate of water flow, sediment particle types, landscape position, size of the vegetated area, and slope.

In conclusion, the soil and water movement part of the externalities research agenda seems the most open to innovation, following up on the pioneers of a new wave in the issue cycle. The basic methods and models exist to do the job, but the research–policy debate is more complicated than for a 'new' issue. Especially where existing policies have done 'the right thing for the wrong reasons' (protecting forests for watershed functions), an over response may be expected when research results lead to a review of established wisdom and lore.

7. Integration at farm and landscape level

From an analytical perspective it is useful to separate the different types of flows, the way their origin is modified by land cover and land use, the way their rate of flow or coupling with substances carried along is modified by filter elements in a landscape and the impact they have on external stakeholders. But in reality certain landscape elements, in particular trees and small patches of forests, can modify a number of flows (e.g. water as well as organisms). Decisions on the management of such filters are based on the trade-offs between positive and negative attributes of these filter elements.

Farmers in north Lampung (Indonesia), for example, readily acknowledge that maintaining a surface mulch provides a sediment filter and reduces soil erosion on slopes, but they are also convinced that it leads to more rats and snakes in their field. Removing all mulch, e.g. by burning crop residues, provides an 'animal filter' that increases their yields and makes the fields safer to work in (Gauthier, 1996).

Farmer knowledge of lateral flows and landscape relations should form a starting point for any effort to understand the rationality of their decisions on landscape elements. Local terminologies for landscape building blocks may contain more information about functional relations than recognised in remotely sensed maps. An example of farmer classification in north Thailand was described by J. Peters (pers. commun.) on the basis of his 2-year participant observation in a Karen village (Table 5). Categories such as 'forest above rice field' do not translate well into English, yet ensure lateral flows into the paddies. As such forests are normally owned by the same family as the paddy, however, and this does not entail an 'externality' in the economic sense. Riparian forests are important for providing *cool* water that is deemed essential for the life of the local spirit owners of the land and the water, e.g. crabs, fish, and frogs that should be found living in a healthy paddy. Other landscape relations and lateral flows in the local knowledge system again refer to the biotic relations of pests in the main food crops. Snakes and the few remaining leopard cats and civets in the still forested landscape are recognised for keeping rat populations under control. In years that the bamboo flowers and sets fruit rats and mice rapidly multiply and the following cropping season rice crops may fail, leading to famine. Lansing et al. (1998) analysed the water temple system in Bali that integrates spatial patterns of rice cultivation in relation to the lateral flows of irrigation water and of pests. Thapa et al. (1995) described a rigorous methodology that is available for further analysis of such local knowledge systems that include lateral flows.

To achieve 'integrated natural resource management' all external effects of land use will somehow have to be taken into account in farmer decision making about the use of natural resources on and off farm.

Table 5
Landscape elements recognised by a Karen community example in the upper Mae Chaem watershed, north Thailand (J. Peters, pers. commun.)

Landscape element	Location	Function	Accessible to	Resource use
Watershed (ridge) forest	On the mountain ridge separating the village territory from the next one	Providing main irrigable water source and clean drinking water (piped to the village)	All	Cattle grazing and collection of food, and medicinal plants, hunting area, NTFP collection
Conservation forest	New category	Conserving wild animals and plants	No hunting	Cattle grazing
Open access forest	Hills surrounding village	Providing forest products	All, with permission	Construction wood (for house, not for sale), grazing and NTFP collection
Community forest	Hills surrounding village, but closer to the village than previous category	Providing forest products, for community activities	Community groups	Wood for community structures, grazing and NTFP collection
Bush fallow (‘revolving forest’)	Closer to the village than previous category	Crop production, grazing land	Privately controlled in cropping years, open access grazing in fallow years	Crop yields, fodder, manure transferred to homegardens, grazing and NTFP collection
Riparian forest	Along the streams and rivers	Providing clean and cool water for irrigation, maintaining the spirit owners (e.g. crabs, fish and frogs) in the paddy fields	All	NTFP’s
‘Forest above paddy field’	Forest land adjacent to a landowner’s paddy field	Reserved for the exclusive use of the paddy owner	Private	Commercial or subsistence gardens or useful tree species
‘Paddy field’	Between streams and previous forest category	Rice production (+dry season vegetable crop)	Private	Rice and dry season crops; cattle/buffalo grazing in dry season
Burial forest	Close to village	Cemetery	All	–
Birth forest	Close to village	Burial of umbilical cords for spiritual security	All	–
Homegarden	Around house in village	Household needs	Private	Fruit, vegetables, fodder, medicine (human and animal)

NTFP: non-timber forest product.

The tradeoff between externalities and private profitability can be essentially different at landscape level than at that of an 'average plot' depending on the spatial pattern of the landscape and the interactions and complementarities it entails. Further incorporation of spatially explicit landscape relations and the role of trees and forest fragments into such models may lead to a more complete evaluation of options to meet the objectives of multiple stakeholders. Yet, a major challenge of such models is to do justice to the large variation among households in the decisions they make. Agrodiversity (the genetic diversity of agriculturally used resources) at landscape scale may be largely determined by the between-farm rather than within-farm diversity, and this diversity is little appreciated in most current approaches to 'priority setting' for research, nor by dissemination and extension activities.

8. Concluding remarks

Recognition of lateral flows as the basis for most, if not all, externalities may lead to the identification of 'keystone elements' in a landscape that have a substantial impact by providing a filter function. Trees and small patches of forest are likely to play a major role in filter functions involving surface or subsurface flows of water and sediments, as well as in the connectivity allowing movement of organisms. Agroforestry research at landscape scale can contribute by the identification of such lateral flows and the opportunities for and limitations of filter functions. Much of the toolbox developed previously for plot level monitoring and modelling has value at the landscape scale as well, but the horizontal dimension that was carefully removed in most experimental approaches of the past (e.g. plots with deep-root trenches or exclusion of incoming run-on and sediment flows) should be reinstated. Lateral flows form an important part of the causal chain in any environmental management issue and it is time that the available methods were used.

Acknowledgements

Thanks are due to J. Peters for introducing us to local landscape knowledge in north Thailand and to many of the 1999 workshop participants for discussions reflected in this paper. Chin Ong, Fergus Sinclair, Mike Swift and Tom Tomich provided thoughtful comments on an earlier version of this manuscript.

References

Anderson, J.M., 1994. Functional attributes of biodiversity in land use systems. In: Greenland, D.J., Szabolcs, I. (Eds.), Soil Resilience and Sustainable Land Management. CAB International, Wallingford, UK, pp. 267–290.

Baffaut, C., Nearing, M.A., Govers, G., 1998. Statistical distributions of soil loss from runoff plots and WEPP model simulations. Soil Sci. Soc. Am. J. 62, 756–763.

Bruijnzeel, L.A., 1990. Hydrology of Moist Tropical Forests and Effects of Conversion: A State of Knowledge Review. UNESCO International Hydrological Programme, Paris, 224 pp.

Bruijnzeel, L.A., 1997. Hydrology of forest plantations in the tropics. In: Nambiarand, E.K.S., Brown, A.G. (Eds.), Management of Soil, Nutrients and Water in Tropical Forest Plantations. ACIAR, Canberra, Australia, pp. 125–167.

Bunnell, F.L., Huggard, D.J., 1999. Biodiversity across spatial and temporal scales: problems and opportunities. For. Ecol. Manage. 115, 113–126.

Burrough, P.A., 1993. Soil variability: a late 20th century view. Soils Fert. 56, 529–562.

Calder, I., 1998. Water resources and land use issues. System-wide Initiative on Water Management, SWIM Paper 3. International Water Management Institute, Colombo, Sri Lanka.

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

Crawford, J.W., Pachepsky, Y.A., Rawls, W.J. (Eds.), 1999. Integrating Processes in Soils Using Fractal Models. Geoderma 88, 3–4 (special issue).

Daily, G.C., Ehrlich, P.R., 1994. Population extinction and the biodiversity crisis. In: Mãler, K.G., Folke, C., Holling, C.S., Jansson, B.O. (Eds.), Biodiversity Conservation: Problems and Policies. Kluwer Academic Publishers, Dordrecht.

Daniels, R.B., Nelson, L.A., 1987. Soil variability and productivity: future developments. In: Boersma, L.L. (Ed.), Future Developments in Soil Science Research. Soil Science Society of America, Madison, WI, USA, pp. 279–293.

Diemont, W.H., Smiet, A.C., Nurdin, 1991. Re-thinking erosion on Java. Neth. J. Agric. Sci. 39, 213–224.

Drake, J.A., Flum, T.E., Witteman, G.J., Voskuil, T., Hoylman, A.M., Creson, C., Kenny, D.A., Huxel, G.R., Larue, C.S., Duncan, J.R., 1993. The construction and assembly of an ecological landscape. J. Anim. Ecol. 62, 117–130.

Ehrlich, P.R., Daily, G.C., 1993. Population extinction and saving biodiversity. Ambio 22, 64–68.

Ericksen, P.J., McSweeney, K., 1999. Fine-scale analysis of soil quality for various land uses and land forms in central Honduras. Am. J. Alternative Agric. 14, 146–157.

Evans, R., 1993. On assessing accelerated erosion of arable land by water. Soils Fert. 56, 1285–1293.
Farina, A., 1998. Principles and Methods in Landscape Ecology. Chapman & Hall, London.
Forman, R.T.T., 1995. Land Mosaics, the Ecology of Landscapes and Regions. Cambridge University Press, Cambridge.
Fryirs, K., Brierley, G.J., 1999. Slope-channel decoupling in Wolumla catchment, New South Wales, Australia: the changing nature of sediment sources following European settlement. Catena 35, 41–63.
Gauthier, R., 1996. Vertebrate pests, crops and soil: the case for an agroforestry approach to agriculture on recently deforested land in North Lampung. AGRIVITA 19, 206–212.
Gerrard, A.J., 1990. Soil variations on hillslopes in humid temperate climates. Geomorphology 3, 225–244.
Grunwald, S., Frede, H.G., 1998. Application of AGNPSm in German watersheds. Wiener Mitteilg. Wasser-Abwasser-Gewaesser-experiences with soil erosion models. Bd. 151, 183–189.
Hamilton, L.S., King, P.N., 1983. Tropical Forested Watersheds: Hydrologic and Soils Response to Major Uses or Conversions. Westview Press, Boulder, CO, USA, 168 pp.
Harrison, S., 1994. Metapopulations and conservation. In: Edwards, P.J., Webb, N.R., May, R.M. (Eds.), Large-scale Ecology and Conservation Biology. Blackwell, Oxford.
Harte, J., Kinzig, A., Green, J., 1999. Self-similarity in the distribution and abundance of species. Science 284, 334–336.
Harvey, C.A., 2000. Windbreaks enhance seed dispersal into agricultural landscapes in Monteverde, Costa Rica. Ecol. Appl. 10, 155–173.
Harvey, C.A., Haber, W.A., 1998. Remnant trees and the conservation of biodiversity in Costa Rican pastures. Agrofor. Syst. 44, 37–68.
Hu, D., Saito, Y., Kempe, S., 1999. Sediment and nutrient transport to the coastal zone. In: Galloway, J.N., Melillo, J.M. (Eds.), Asian Change in the Context of Global Climate Change. Cambridge University Press, Cambridge, UK, pp. 245–270.
Hubbell, S.P., 2001. The Unified Neutral Theory of Biodiversity and Biogeography. Monographs in Population Biology 32. Princeton University Press, Princeton.
Iida, T., 1999. A stochastic hydro-geomorphological model for shallow landsliding due to rainstorm. Catena 34, 293–313.
Izac, A.M., Sanchez, P.A., 1999. Towards a natural resource management paradigm for international agriculture: example of agroforestry research. Agric. Syst. 69, 5–25.
Kabrick, J.M., Clayton, M.K., McBratney, A.B., McSweeney, K., 1997. Cradle-knoll patterns and characteristics on drumlins in Northeastern Wisconsin. Soil Sci. Soc. Am. J. 61, 595–603.
Kramer, R., Van Schaik, C., Johnson, J. (Eds.), 1997. Last Stand: Protected Areas and the Defense of Tropical Biodiversity. Oxford University Press, New York, 242 pp.
Kunin, W.E., 1998. Extrapolating species abundance across spatial scales. Science 281, 1513–1515.
Lam, N.S.N., de Cola, L., 1993. Fractals in Geography. PTR Prentice-Hall, Englewood Cliffs, NJ, USA, 308 pp.
Lansing, J.S., Kremer, J.N., Smuts, B.B., 1998. System-dependent selection, ecological feedback and the emergence of functional structure in ecosystems. J. Theor. Biol. 192, 377–391.
Lowrance, R., 1998. Riparian ecosystems as filters for nonpoint-source pollution. In: Pace, M.L., Groffman, P.M. (Eds.), Successes, Limitations and Frontiers in Ecosystem Science. Springer, Berlin, pp. 113–141.
Malingreau, J.P., Zhuang, Y.H., 1999. Biomass burning: an ecosystem process of global significance. In: Galloway, J.N., Melillo, J.M. (Eds.), Asian Change in the Context of Global Climate Change. Cambridge University Press, Cambridge, UK, pp. 101–127.
Meybeck, M.P., Green, P., Vororsmarty, C.J., 2001. A new typology for mountains and other relief classes: an application to global continental water resources and population distribution. Mountain Res. Dev. 21, 34–45.
Moore, I.D., Gessler, P.E., Nielsen, G.A., Peterson, G.A., 1993a. Soil attribute prediction using terrain analysis. Soil Sci. Soc. Am. J. 57, 443–452.
Moore, I., Turner, A.K., Wilson, J.P., Jenson, S.K., Band, L.E., 1993b. GIS and land-surface-subsurface process modeling. In: Goodchild, M.F., et al. (Eds.), Environmental Modeling and GIS. Oxford University Press, Oxford, UK, pp. 197–230.
Noble, I.R., 1999. Effect of landscape fragmentation, disturbance and succession on ecosystem function. In: Tenhunen, J.D., Kabat, P. (Eds.), Integrating Hydrology, Ecosystem Dynamics and Biogeochemistry in Complex Landscapes. Wiley, Chichester, UK, pp. 297–312.
Opdam, P., Van Apeldoorn, R., Schotman, A., Kalkhoven, J., 1993. Population responses to landscape fragmentation. In: Vos, C.C., Opdam, P. (Eds.), Landscape Ecology of a Stressed Environment. IALE Studies in Landscape Ecology No. 1. Chapman & Hall, London, pp. 147–171.
Pennock, D.J., Anderson, D.W., de Jong, E., 1994. Landscape-scale changes in indicators of soil quality due to cultivation in Saskatchewan, Canada. Geoderma 64, 1–19.
Risser, P.G., 1989. The movement of nutrients across heterogenous landscapes. In: Clarholm, M., Bergstrom, L. (Eds.), Ecology of Arable Land. Kluwer Academic Publishers, Dordrecht, pp. 247–251.
Rosenzweig, M.L., 1995. Species Diversity in Space and Time. Cambridge University Press, Cambridge, UK, 436 pp.
Rosenzweig, M.L., 1999. Heeding the warning in biodiversity's basic law. Science 284, 276–277.
Simberloff, D., 1999. The role of science in the preservation of forest biodiversity. For. Ecol. Manage. 115, 101–111.
Sinclair, F.L., 1999a. A general classification of agroforestry practice. Agrofor. Syst. 46, 161–180.
Sinclair, F.L., 1999b. The agroforestry concept-managing complexity. Scot. For. 53, 12–17.
Stocking, M., 1998. Measuring and assessing the impacts of erosion: the particular challenges on communal rangelands in Africa. Paper presented at the SACRAN Communal Rangelands Symposium, Fort Hare, 6–9 July, 1998. Available from: www.uea.ac.uk/dev/publink/stocking/ms98c.shtml.
Thapa, B., Sinclair, F.L., Walker, D.H., 1995. Incorporation of indigenous knowledge and perspectives in agroforestry development. Part Two. Case-study on the impact of explicit representation of farmers' knowledge. Agrofor. Syst. 30, 249–261.

Tomich, T.P., Fagi, A.M., de Foresta, H., Michon, G., Murdiyarso, D., Stolle, F., van Noordwijk, M., 1998. Indonesia's fires: smoke as a problem, smoke as a symptom. Agrofor. Today 10, 4–7.

Tongway, D.J., 1990. Soil and landscape processes in the restoration of rangelands. Aust. Rangeland J. 12, 54–57.

Tongway, D.J., 1994. Rangeland Soil Condition Assessment Manual. CSIRO Publications, Australia.

Turchin, P., 1998. Quantitative Analysis of Movement: Measuring and Modeling Population Redistribution in Animals and Plants. Sinauer Associates, Sunderland, MA, USA, 396 pp.

Vandermeer, J., Van Noordwijk, M., Ong, C., Anderson, J., Perfecto, Y., 1998. Global change and multi-species agroecosystems: concepts and issues. Agric. Ecosyst. Environ. 67, 1–22.

Van Noordwijk, M., 1999a. Nutrient cycling in ecosystems versus nutrient budgets of agricultural systems. In: Smaling, E., Oenema, O., Fresco, L. (Eds.), Nutrient Cycles and Nutrient Budgets in Global Agro-ecosystems. CAB International, Wallingford, pp. 1–26.

Van Noordwijk, M., 1999b. Scale effects in crop fallow rotations. Agrofor. Syst. 47, 239–251.

Van Noordwijk, M., Ong, C.K., 1996. Lateral resource flow and capture—the key to scaling up agroforestry results. Agrofor. Forum 7, 29–31.

Van Noordwijk, M., Van Roode, M., McCallie, E.L., Lusiana, B., 1998. Erosion and sedimentation as multiscale, fractal processes: implications for models, experiments and the real world. In: Penning de Vries, F., Agus, F., Kerr, J. (Eds.), Soil Erosion at Multiple Scales, Principles and Methods for Assessing Causes and Impacts. CAB International, Wallingford, pp. 223–253.

Vos, C.C., Opdam, P. (Eds.), 1993. Landscape Ecology of a Stressed Environment. Chapman & Hall, London, 310 pp.

Watson, A., Evans, R., 1991. A comparison of estimates of soil erosion made in the field and from photographs. Soil Till. Res. 19, 17–27.

Wenzel, W.W., Unterfrauner, H., Schulte, A., Ruhiyat, D., Simonrangkir, D., Kuraz, V., Bradstetter, A., Blum, W.E.H., 1998. Hydrology of acrisols beneath Dipterocarp forests and plantations in East Kalimantan, Indonesia. In: Schulte, A., Ruhiyat, D. (Eds.), Soils of Tropical Forest Ecosystems. Springer, Berlin, pp. 62–72.

Whelan, R.J., 1995. The Ecology of Fire. Cambridge University Press, Cambridge, UK, 346 pp.

Wibowo, A., Suharti, M., Sagala, A.P.S., Hibani, H., Van Noordwijk, M., 1997. Dealing with fire on *Imperata* grasslands as part of agroforestry development in Indonesia. Agrofor. Syst. 36, 203–217.

Wood, E.F., 1999. The role of lateral flow: over- or underrated. In: Tenhunen, J.D., Kabat, P. (Eds.), Integrating Hydrology, Ecosystem Dynamics and Biogeochemistry in Complex Landscapes. Wiley, Chichester, UK, pp. 197–215.

Available online at www.sciencedirect.com
SCIENCE DIRECT®

Agriculture, Ecosystems and Environment 104 (2004) 63–73

Agriculture Ecosystems & Environment

www.elsevier.com/locate/agee

Terrestrial pteridophytes as indicators of a forest-like environment in rubber production systems in the lowlands of Jambi, Sumatra

H. Beukema[a,*], M. van Noordwijk[b]

[a] *Department of Plant Biology, RUG Biological Sciences, University of Groningen, P.O. Box 14, 9750 AA Haren, The Netherlands*
[b] *International Centre for Research in Agroforestry (ICRAF) S.E. Asia, P.O. Box 161, Bogor 16001, Indonesia*

Abstract

Species richness of terrestrial ferns and fern allies (Pteridophyta) may indicate forest habitat quality, as analysed here for a tropical lowland area in Sumatra. A total of 51 standard 0.16 ha plots in primary forest, rubber (*Hevea brasiliensis*) agroforests and rubber plantations was compared for plot level diversity (average number of species per plot) and landscape level diversity (species–area curves). Average plot level species richness (11 species) was not significantly different amongst the three land use types. However at the landscape level the species–area curve for rubber agroforests (also called jungle rubber) had a significantly higher slope parameter than the curve for rubber plantations, indicating higher beta diversity in jungle rubber as compared to rubber plantations. Plot level species richness is thus not fully indicative of the (relative) richness of a land use type at the landscape scale because scaling relations differ between land use types. Terrestrial fern species can serve as indicators of disturbance or forest quality as many species show clear habitat differentiation with regard to light conditions and/or humidity. To assess forest habitat quality in rubber production systems as compared to primary forest, terrestrial pteridophyte species were grouped according to their ecological requirements into 'forest species' and 'non-forest species'. Species–area curves based on 'forest species' alone show that the understorey environment of jungle rubber supports intermediate numbers of 'forest species' and is much more forest-like than that of rubber plantations, but less than primary forest. Species richness alone, without a priori ecological knowledge of the species involved, did not provide this information. Jungle rubber systems can play a role in conservation of part of the primary rain forest species, especially in areas where the primary forest has already disappeared. In places where primary forest is gone, jungle rubber can conserve part of the primary forest species, but large areas of jungle rubber are needed. In places where primary forest is still present, priority should be given to conservation of remaining primary forest patches.

Keywords: Agroforestry; Biodiversity; Jungle rubber; Pteridophyta; Species–area relations; Tropical rain forest

1. Introduction

With the disappearance of undisturbed lowland rain forest habitat the question arises whether disturbed habitat maintains some of the characteristics and functions of the original forest, to what extent it can support survival and reproduction of primary rain forest species and how this function is influenced by management practices. For a complete answer of this question we would have to consider all major taxa of flora and associated fauna. The research reported here compares diversity of terrestrial pteridophyte species, with known habitat requirements, to assess for this group to what extent the understorey habitat in rubber produc-

* Corresponding author. Tel.: +31-50-363-2281;
fax: +31-50-363-2273.
E-mail address: h.beukema@biol.rug.nl (H. Beukema).

0167-8809/$ – see front matter © 2004 Elsevier B.V. All rights reserved.
doi:10.1016/j.agee.2004.01.007

tion systems is comparable to the understorey habitat in undisturbed rain forest for the lowland peneplain of Jambi (Sumatra).

1.1. Exploratory research and remaining questions

De Foresta and co-workers were probably the first to study the vegetation of rubber agroforests (also called 'jungle rubber'; Gouyon et al., 1993) to get an impression of species richness. Sampling a 100 m transect line (Michon and De Foresta, 1995) they found almost twice as many herb species in a rubber agroforest as compared to a nearby primary forest (23 versus 12 species) in Jambi Province, Sumatra. Their research was broad in the sense that all vegetation was included, but limited in the fact that vegetation types were represented by a 100 m transect only and that the study was not replicated across the landscape. When a larger number of plots are sampled, will the average number of herb species per plot remain twice as high for jungle rubber as compared to primary forest? Another question that remained after the exploratory work by Michon and De Foresta was whether high diversity found on the plot level is a reflection of high species turnover (beta diversity) on a landscape scale, or not. Data on plot level have been used (Leakey, 1999) to make statements that 'complex, multistrata agroforests contain about 70% of all the regional pool of plant species', apparently assuming that a single transect line is sufficient to characterise a vegetation type and that scaling rules above plot level do not differ between vegetation types.

1.2. Species turnover and species composition

In spite of a high number of species found at the plot level, if the species composition in jungle rubber at the landscape level would be rather repetitive, in other words if the species–area curve for jungle rubber would have a much lower slope parameter than the curve for primary forest, those rubber agroforests would probably not be as interesting an option for biodiversity conservation.

Species richness, regardless of species composition, is often used as a measure in biodiversity studies. If we deal with disturbed ecosystems however, there are risks involved because different taxa react in different ways to disturbance. For many taxa, "diversities peak at intermediate rates of small-scale disturbance" (Rosenzweig, 1995, p. 39). Although species are considered the 'currency' of biodiversity, counting just any species does not help us much when we are interested in conservation of a specific ecosystem. What kind of species do we find? Do the species we find give us some information about the quality of the type of habitat we are interested in? The fact that we can find great diversity of pteridophyte species on the forest floor of rubber agroforests does not tell us that the environment there is comparable to a primary forest and can be expected to support primary forest species.

1.3. Terrestrial pteridophytes as an indicator group

For assessments using an indicator group we should know first of all whether the group of species we are using contains enough species that differ in habitat requirements with respect to the range of the environmental factors that change when a forest is disturbed by human action. If the great majority of pteridophyte species were generalist species that could grow anywhere they would not indicate any changes in forest environment due to disturbance. Enough species with narrow habitat requirements are needed so they can be grouped to indicate different degrees of disturbance. Important environmental factors for life in the understorey of a tropical lowland rain forest that change with disturbance are light conditions (quantity and spectrum) and microclimate (moisture and temperature regime). When species are thus grouped we can assess which part of the total diversity in each land use type is made up by species requiring forest-like conditions, assuming that the bigger the share of those 'forest species', the more forest-like the understorey environment will tend to be.

1.4. Research questions

Summarising the above, the research is focussed on the following questions:

- Can rubber production systems play a role in conservation of primary forest species by providing forest-like habitat?
- Can terrestrial pteridophyte species indicate disturbance level or habitat quality of the forest understorey?

- Is plot level species richness indicative of the (relative) richness of a land use type at the landscape scale, or do scaling relations differ essentially between land use types?
- Is species richness a useful indicator of habitat quality, or is (a priori) ecological information needed on the species involved?

2. Land use change in the Jambi lowlands

The study was carried out in the lowlands of the peneplain area in Jambi Province, Sumatra at elevations of 40–150 m above sea level. For sampling locations see Fig. 1.

The original forests of this area are mixed Dipterocarp rain forests. The physical environment, structure and floristics of these forests and of the derived secondary vegetation types are described by Laumonier (Laumonier, 1997, pp. 88–130). Extensive research on land use and land use changes has been carried out by the 'Alternatives to Slash-and-Burn' project and summarised in two reports (Van Noordwijk et al., 1995; Tomich et al., 1998). Land use types described by the ASB project (Tomich et al., 1998, Table I.2, p. 19) include natural forest, forest extraction (community-based forest management, commercial logging), complex multistrata agroforestry systems (rubber agroforests), simple tree crop systems (rubber, oil palm (*Elaeis guineensis*) and industrial timber monoculture), crop/fallow systems (upland rice (*Oryza sativa*)/bush fallow rotation), continuous annual cropping systems (monoculture cassava degrading to *Imperata cylindrica*), and grasslands/pasture (*I. cylindrica*).

Primary and logged-over forests in the Jambi lowlands are disappearing fast in recent years, they are replaced mainly by plantations (oil palm, rubber, timber) and to a lesser extent by smallholder agroforests (rubber, fruit trees). By the end of the 1990s much of the lowland primary and logged-over forests as shown on Laumonier's 1986 vegetation map (Laumonier, 1997) had already been converted to other land uses (survey by H. Beukema, 1997). Unfortunately an up to date land use map showing these current rapid changes is not available. For generalised maps of land use changes in the Jambi lowlands in the 1980s, see Beukema et al. (1997).

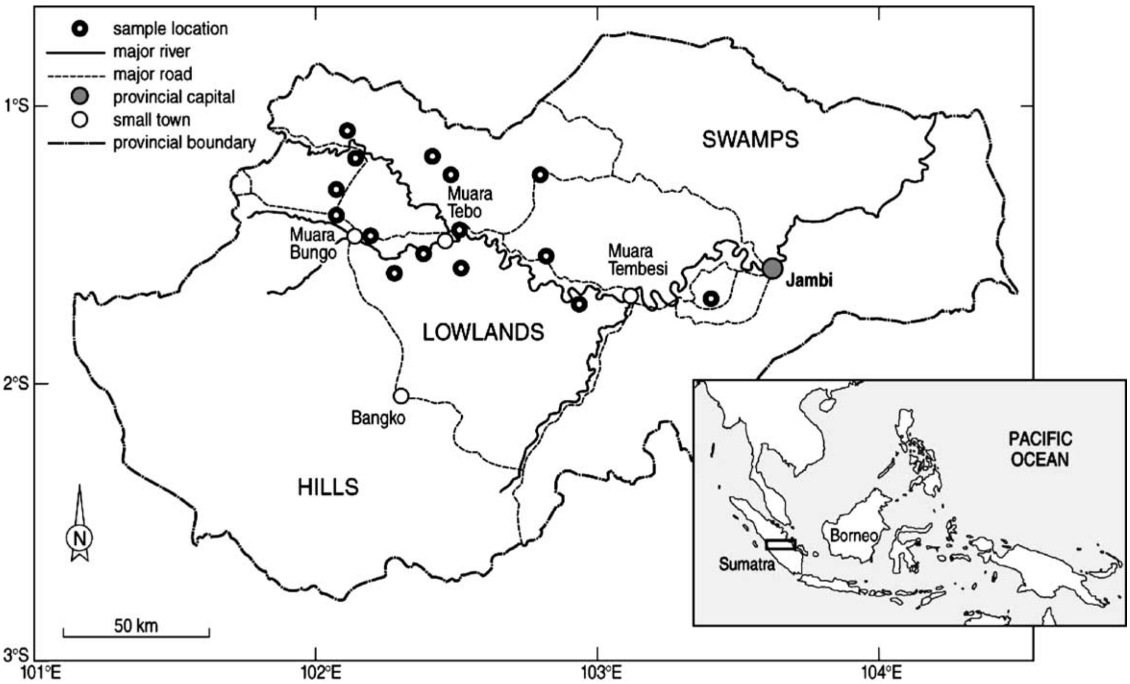

Fig. 1. Sampling locations in the lowland area of Jambi Province, Sumatra.

3. Rubber production systems

In Jambi Province rubber is produced mainly in rubber agroforests and to a lesser extent in more intensively managed monocultural plantations. Both production systems use slash-and-burn to clear land before planting.

In the monocultural plantations rubber (latex) is the only product. The undergrowth below the rubber trees is kept low by using herbicides and by manual weeding, while fertilisers are applied around the rubber trees to stimulate their growth. Tapping starts when the rubber trees are 5–6 years old. Trees remain productive until they are 20–25 years old and a new planting cycle starts.

In the jungle rubber production system there are a number of secondary products next to rubber (latex) that is the main product. Rubber is planted together with rice, vegetables, herbs, and a limited number of useful trees such as fruit trees. Weeds are controlled manually and only during the first 2 or 3 years when rice and vegetables are produced. After that the secondary vegetation that comes in naturally and includes useful species is allowed to grow with the rubber. A dense secondary forest vegetation builds up. Around 9 years after planting, a path between the rubber trees is made and tapping starts. Through natural regeneration of rubber seedlings and active replanting in gaps by the farmer (Wibawa et al., in review), those rubber agroforests can remain productive much longer than rubber plantations. A secondary forest dominated by rubber is the result. In an average 'jungle rubber' agroforest only about 40% of trees with a diameter at breast height (DBH) of over 10 cm are rubber trees. The other trees are mostly natural regrowth while some trees are planted by the farmer.

4. Method

4.1. Plot sampling

Three land use types with associated anthropogenic disturbance levels were sampled: undisturbed rain forest (11 plots), low disturbance jungle rubber (23 plots) and high disturbance rubber plantations (17 plots). The 'undisturbed' rain forest was old growth forest without visible traces of timber cutting and without known history of logging or shifting cultivation, the only human use being limited collection of non-timber forest products and hunting.

Plots were located across the Jambi peneplain, a slightly undulating to flat area of around 200 km × 150 km with rather uniform soils in the centre of Sumatra. The total area of each land use type in the Jambi peneplain is unknown, but the area under jungle rubber is much larger than the area under either rubber plantation or undisturbed forest. In each land use type, the total area sampled is very small compared to the total area of the land use type, so the differences in sampling intensity are probably less important.

Standard plots of 40 m × 40 m (0.16 ha per plot) were subdivided into 16 subplots of 10 m × 10 m each. Counting presence of terrestrial pteridophyte species in the 16 subplots of each plot resulted in a frequency score between 0 and 16 for each species in each plot. For this paper, only presence of species in plots was analysed. Edge effects were avoided by locating the plots away from forest edges and roadsides. Small paths used by rubber tappers however were considered characteristic of jungle rubber systems and therefore not avoided. Plots were located well away from rivers and streams to avoid rheophytes that indicate moisture rather than any level of anthropogenic disturbance.

Only productive rubber systems were sampled. Age of jungle rubber plots varied from 9 to 74 years, while the age of rubber plantation plots was 5–19 years old.

4.2. Pteridophyte grouping

Pteridophyte species were grouped based on ecological notes in literature on Malaysian species (Alston, 1937; Backer and Posthumus, 1939; Fletcher and Kirkwood, 1979; Holttum, 1932, 1938, 1959a,b, 1963, 1966, 1974, 1981, 1991; Holttum and Hennipman, 1978; Kramer, 1971; Page, 1976; Pemberton and Ferriter, 1998; Piggot and Piggot, 1988; Spicer et al., 1985; Wong, 1982). From the literature, it became clear that there is enough habitat differentiation among species to make pteridophytes potentially a suitable indicator group for this study. We would have liked to classify our species by their optima for both light and microclimate conditions, but the available species descriptions (mostly from taxonomical literature) included consistent information on light requirements and preferred habitat only. Nevertheless that information was sufficient to classify the species into ecological groups for the purpose of this study. Based on the literature four levels for light conditions were distinguished: ‘open’ conditions, ‘open/light shade’, ‘light shade’ and ‘shade/deep shade’. In combination with data on preferred habitat the species were assigned to one of two groups arbitrarily named ‘forest species’ and ‘non-forest species’.

‘Forest species’ are all species that require shade or deep shade plus the species that require light shade and grow in forest. ‘Non-forest species’ are all species of open and open/light shade conditions plus the species that require light shade and prefer habitats other than forest (roadsides, forest edges, plantations, etc.). This grouping does not imply that ‘non-forest species’ never grow in the forest. Some of them do occur in forest, especially in gaps, but they are more abundant in open conditions. Species are thus grouped by (inferred) ecological optimum rather than by ecological range.

Of a total of 65 terrestrial pteridophyte species found in the survey, 36 were classified as ‘forest species’ and 26 as ‘non-forest species’ (see Table 1).

Three species remained unclassified because they were not identified to the species level and could not be linked to literature (see Table 1). They were excluded from analyses concerning ‘forest species’. Although species–area curves are of course sensitive to the removal of species from the data, we expect the effects to be limited in this case. Of the three species that were excluded, two unclassified *Cyathea* species (labelled *Cyathea* sp.2 and *Cyathea* sp.3) were most likely not ‘forest species’ in our classification and would not have been included in the analysis anyway. They were not encountered in forest at all. *Cyathea* sp.2 occurred more often in rubber plantations than in jungle rubber: it was found in four rubber plantation plots and in one jungle rubber plot (24 and 4% of those plots, respectively) while *Cyathea* sp.3 occurred in one rubber plantation plot and in one jungle rubber plot. Both species were found to be growing more abundantly in the rubber plantation plots than in the jungle rubber plots. The third species that was excluded was an unclassified *Asplenium* species occurring as a single individual in a jungle rubber plot.

4.3. Data analysis

For statistical analysis the program SPSS Version 10.0 was used.

At the plot level, differences between land use types for average number of (forest) species per plot were tested using one-way ANOVA and Tukey’s HSD test.

At the landscape level, to analyse species–area relations the program EstimateS (Colwell, 1997) was used to randomise plot sequence 100,000 times for each land use type and derive average cumulative richness values.

A logarithmic equation of the form:

$$y = b \ln x + a \tag{1}$$

was fitted through the resulting points, where y is the cumulative number of species, b the scaling relation of species richness (beta diversity), x the cumulative number of 0.16 ha plots (area), and a a constant estimating the average richness for a single plot (alpha diversity).

The ‘area’ in the species–area curves represents a collection of non-adjacent 0.16 ha plots scattered over a vast landscape.

The distances between plots are comparable for forest and jungle rubber: the average distance between plots was for forest plots 42 km (S.E. = 3.6) and for jungle rubber plots 39 km (S.E. = 1.5). Non-parametric tests show that also the distributions of interplot distances are comparable for forest and jungle rubber. However, the interplot distances of the rubber plantation plots were different both in average (as high as 74 km, S.E. = 5.2) and in distribution. This is due to the fact that there are only two large rubber estates in the Jambi lowlands that have rubber trees of the higher age classes that we needed to include in the sampling, and those two estates are far apart (one near Muara Bungo, the other near Jambi town). As a result, long distances are over represented in the rubber plantation sample. This may have caused a slight overestimation of the slope parameters of the species–area curves for rubber plantations, but such overestimation would not seriously affect our main conclusions.

The slope parameters (b) found for the three land use types were compared statistically by linear regression over their common area range of 11 plots (1.76 ha).

Table 1
Species list of terrestrial pteridophyte species found in Jambi lowlands, for classification criteria see text

Family	Species name	Group
Aspleniaceae	*Asplenium glaucophyllum* v.A.v.R.	Non-forest
Aspleniaceae	*Asplenium longissimum* Bl.	Non-forest
Aspleniaceae	*Asplenium pellucidum* Lam.	Forest
Aspleniaceae	*Asplenium* sp.	Not classified
Blechnaceae	*Blechnum finlaysonianum* Hk. & Grev.	Forest
Blechnaceae	*Blechnum orientale* L.	Non-forest
Blechnaceae	*Stenochlaena palustris* (Burm.) Bedd.	Non-forest
Cyatheaceae	*Cyathea* cf. *contaminans* (Hooker) Copel.	Non-forest
Cyatheaceae	*Cyathea moluccana* R. Br.	Forest
Cyatheaceae	*Cyathea* sp.2	Not classified
Cyatheaceae	*Cyathea* sp.3	Not classified
Dennstaedtiaceae	*Lindsaea* cf. *repens* (Bory) Thw.	Forest
Dennstaedtiaceae	*Lindsaea cultrata* (Willd.) Swartz	Forest
Dennstaedtiaceae	*Lindsaea divergens* Hk. & Grev.	Forest
Dennstaedtiaceae	*Lindsaea doryphora* Kramer	Forest
Dennstaedtiaceae	*Lindsaea ensifolia* Swartz	Non-forest
Dennstaedtiaceae	*Lindsaea parasitica* (Roxb. Ex Griffith) Hieron.	Forest
Dennstaedtiaceae	*Microlepia speluncae* (L.) Moore	Non-forest
Dennstaedtiaceae	*Pteridium caudatum* (L.) Maxon subsp. yarrabense (Domin) Parris	Non-forest
Dryopteridaceae	*Diplazium crenatoserratum* (Bl.) Moore	Forest
Dryopteridaceae	*Diplazium malaccense* C. Presl	Forest
Dryopteridaceae	*Diplazium pallidum* Bl.	Forest
Dryopteridaceae	*Diplazium riparium* Holtt.	Forest
Dryopteridaceae	*Diplazium tomentosum* Bl.	Forest
Dryopteridaceae	*Pleocnemia irregularis* (C. Presl) Holtt.	Forest
Dryopteridaceae	*Tectaria barberi* (Hk.) Copel.	Forest
Dryopteridaceae	*Tectaria fissa* (Kunze) Holtt.	Forest
Dryopteridaceae	*Tectaria singaporeana* (Wall. ex Hk. & Gr.) Copel.	Forest
Dryopteridaceae	*Tectaria vasta* (Bl.) Copel.	Forest
Gleicheniaceae	*Dicranopteris linearis* (Burm. f.) Underw. var. linearis	Non-forest
Gleicheniaceae	*Dicranopteris linearis* (Burm. f.) Underw. var. subpectinata (Christ.) Holtt.	Non-forest
Hymenophyllaceae	*Trichomanes javanicum/singaporeanum*	Forest
Hymenophyllaceae	*Trichomanes obscurum* Bl.	Forest
Lomariopsidaceae	*Teratophyllum* cf. *ludens* (Fée) Holtt.	Forest
Lomariopsidaceae	*Teratophyllum* cf. *rotundifoliatum* (R. Bonap.) Holtt.	Forest
Lycopodiaceae	*Lycopodium cernuum* L.	Non-forest
Nephrolepidaceae	*Nephrolepis biserrata* (Sw.) Schott	Non-forest
Ophioglossaceae	*Helminthostachys zeylanica* L. Hook.	Non-forest
Ophioglossaceae	*Ophioglossum reticulatum* L.	Non-forest
Polypodiaceae	*Microsorum scolopendria* (Burm. f.) Copel.	Non-forest
Pteridaceae	*Adiantum latifolium* Lam.	Non-forest
Pteridaceae	*Pityrogramma calomelanos* (L.) Link	Non-forest
Pteridaceae	*Taenitis blechnoides* (Willd.) Sw.	Forest
Schizaeaceae	*Lygodium circinnatum* (Burm. f.) Sw.	Forest
Schizaeaceae	*Lygodium flexuosum* (L.) Sw.	Non-forest
Schizaeaceae	*Lygodium longifolium* (Willd.) Sw.	Non-forest
Schizaeaceae	*Lygodium microphyllum* (Cav.) R.Br.	Non-forest
Schizaeaceae	*Lygodium salicifolium* Presl	Non-forest
Schizaeaceae	*Schizaea dichotoma* (L.) Sm.	Forest
Schizaeaceae	*Schizaea digitata* (L.) Sw.	Forest
Selaginellaceae	*Selaginella caulescens* (Wall.) Spring	Forest
Selaginellaceae	*Selaginella intermedia* (Bl.) Spring	Forest
Selaginellaceae	*Selaginella plana* (Desv.) Hieron.	Forest

Table 1 (*Continued*)

Family	Species name	Group
Selaginellaceae	*Selaginella roxburghii* (Hk. & Gr.) Spring	Forest
Selaginellaceae	*Selaginella willdenowii* (Desv.) Baker	Non-forest
Thelypteridaceae	*Amphineuron* sp.	Non-forest
Thelypteridaceae	*Christella parasitica* (L.) Lév.	Non-forest
Thelypteridaceae	*Christella subpubescens* (Bl.) Holtt.	Non-forest
Thelypteridaceae	*Mesophlebion chlamydophorum* (C.Chr.) Holtt.	Forest
Thelypteridaceae	*Mesophlebion motleyanum* (Hook.) Holtt.	Forest
Thelypteridaceae	*Pronephrium glandulosum* (Bl.) Holtt.	Forest
Thelypteridaceae	*Pronephrium rubicundum* (v.A.v.R.) Holtt.	Forest
Thelypteridaceae	*Pronephrium* sp.	Forest
Thelypteridaceae	*Pronephrium triphyllum* (Sw.) Holtt.	Non-forest
Thelypteridaceae	*Sphaerostephanos heterocarpus* (Bl.) Holtt.	Forest

Families according to Kubitzki (1990).

5. Results

5.1. Plot level results

The average number of terrestrial pteridophyte species per plot in the current study was indeed higher in jungle rubber (on average 11.7 species) than in primary forest (on average 9.4 species), but not twice as high as found by Michon and De Foresta for herbs, and the difference found is not statistically significant.

Applying a priori ecological knowledge about our species, we find that the plot level species richness in jungle rubber and in rubber plantations is largely due to an increase in species that have their optima in environments other than the shady forest understorey, in our classification 'non-forest species'. Fig. 2 shows the differences in average number of species per plot for the three land use types. Differences are small and not statistically significant when all species are considered ($F[2, 48] = 1.846$, $P = 0.169$), while those differences are large and statistically significant when only 'forest species' are considered ($F[2, 48] = 18.112$, $P < 0.0005$; Table 2).

5.2. Landscape level results, all species

Looking at the landscape level, we see that the species–area curves for pteridophytes in primary forest, jungle rubber and rubber plantations are close together (Fig. 3).

We tested for equality of slopes of the regressions for the three land use types, and found that including interactions (which allows for different slopes) significantly improved the model ($F[2, 27] = 4.005$, $P = 0.030$). The slope parameter of the jungle rubber land use type was significantly higher than the slope parameter of the rubber plantations land use type ($t = 2.827$, $P = 0.009$). The slope parameter of the forest was not significantly different from the slope parameters of the jungle rubber land use type and the rubber plantations land use type ($t = -1.534$, $P = 0.137$

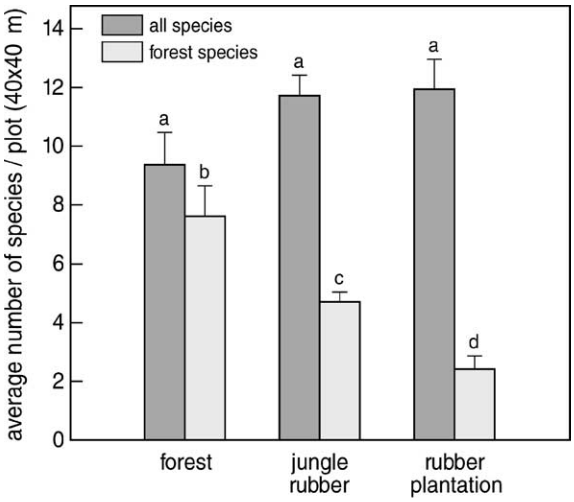

Fig. 2. Number of terrestrial pteridophyte species per 0.16 ha plot. Means and their standard errors for three land use types: forest ($n = 11$), jungle rubber ($n = 23$) and rubber plantations ($n = 17$). Dark bars: all data; light bars: 'forest species' subset. Different letters indicate significant differences between land use types (Tukey's HSD test, $P < 0.05$, see Table 2).

Table 2
Analysis of variance and post-hoc multiple comparisons for data in Fig. 2: number of species per plot (all species, 'forest species')

	N	Mean of all species per plot	S.E. of the mean	Mean of 'forest species' per plot	S.E. of the mean
Forest	11	9.4	1.08	7.6	1.06
Jungle rubber	23	11.7	0.70	4.7	0.36
Rubber plantations	17	11.9	0.99	2.4	0.41
All land use types	51	11.2	0.52		

ANOVA

	Sum of squares	d.f.	Mean square	*F*	Significance
Number of terrestrial pteridophyte species					
Between groups	49.649	2	24.824	1.846	0.169
Within groups	645.528	48	13.448		
Total	695.176	50			
Number of 'forest species'					
Between groups	176.644	2	88.322	18.112	0.000
Within groups	234.062	48	4.876		
Total	410.706	50			

Multiple comparisons (Tukey's HSD). Dependent variable: number of 'forest species'

Land use (*I*)	Land use (*J*)	Mean difference ($I - J$)	S.E.	Significance	95% confidence interval	
					Lower bound	Upper bound
Primary forest	Jungle rubber	2.89*	0.81	0.002	0.94	4.85
	Rubber plantation	5.13*	0.85	0.000	3.07	7.20
Jungle rubber	Primary forest	−2.89*	0.81	0.002	−4.85	−0.94
	Rubber plantation	2.24*	0.71	0.007	0.53	3.95
Rubber plantation	Primary forest	−5.13*	0.85	0.000	−7.20	−3.07
	Jungle rubber	−2.24*	0.71	0.007	−3.95	−0.53

* The mean difference is significant at the 0.05 level.

and $t = 1.292$, $P = 0.207$, respectively). Figs. 2 and 3 and the statistical testing make clear that the pattern at the plot scale is not reflected at the landscape scale. Jungle rubber shows higher beta diversity for terrestrial pteridophytes at the landscape scale than rubber plantations, despite similar plot level diversity.

5.3. Landscape level results, 'forest species'

After grouping species into 'forest species' and 'non-forest species' a second set of species–area curves was constructed based only on 'forest species'. These curves for 'forest species' (Fig. 4) show the part of the total diversity in each land use type (as in Fig. 3) that consists of species that prefer conditions prevalent in undisturbed forest.

Slopes of the regression lines for 'forest species' (Table 3) differ significantly ($F[2, 27] = 352.161$, $P < 0.0005$). The regression line for forest has a steeper slope than the regression lines for jungle rubber and rubber plantations ($t = 17.544$, $P < 0.0005$ and $t = 26.017$, $P < 0.0005$, respectively), and the regression line for jungle rubber has a steeper slope than the regression line for rubber plantations ($t = 8.473$, $P < 0.0005$). The differences between the curves for primary forest (upper line), jungle rubber (middle line) and rubber plantations (lower line) show that the understorey environment of jungle rubber is much more forest-like than that of rubber plantations, but less than primary forest.

The number of jungle rubber plots added up to find the same number of 'forest species' in jungle rubber as in primary forest is progressively larger at the higher

Table 3
Slopes and their standard errors for the species–area regressions (all species, 'forest species') in Figs. 3 and 4

	N	Slope parameter all species	S.E. of the slope	Slope parameter 'forest species'	S.E. of the slope
Forest	11	9.71	0.16	8.34	0.14
Jungle rubber	11	10.11	0.21	5.07	0.11
Rubber plantations	11	9.37	0.18	3.49	0.14

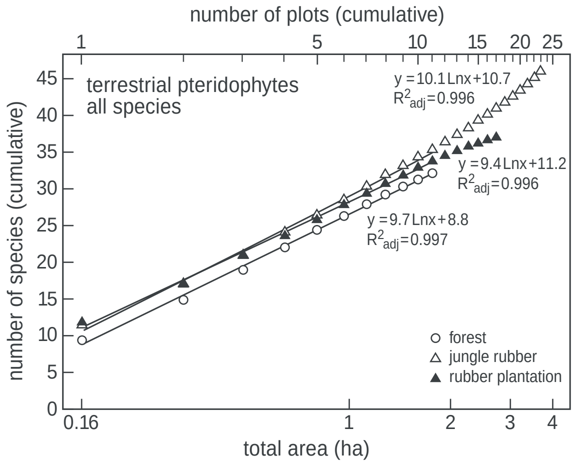

Fig. 3. Species–area curves for terrestrial pteridophytes in forest, jungle rubber and rubber plantations. Plots were 0.16 ha each, non-adjacent and spread over a large area (see Fig. 1). Plots were randomised 100,000 times to remove the effect of plot order.

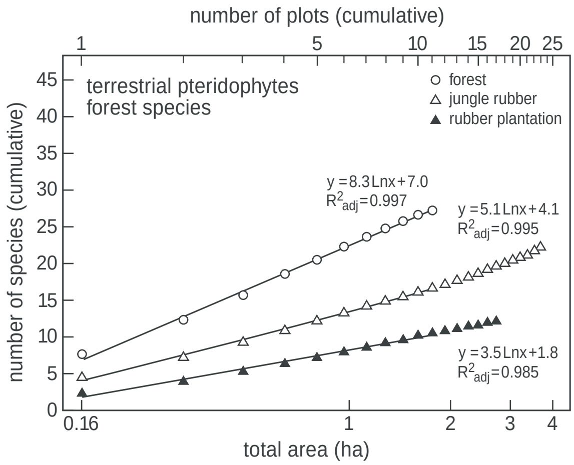

Fig. 4. Species–area curves for 'forest species' subset of terrestrial pteridophytes in forest, jungle rubber and rubber plantations. Plots were 0.16 ha each, non-adjacent and spread over a large area (see Fig. 1). Plots were randomised 100,000 times to remove the effect of plot order.

levels of species richness associated with larger areas. When S represents the number of 'forest species', we find at $S = 15$ we need 3.0 jungle rubber plots for each primary forest plot, at $S = 20$ we need 4.0 and at $S = 25$ we would need 5.3 jungle rubber plots for each primary forest plot.

In addition to the differences in diversity of 'forest species', our data show that some of the 'forest species' that are found in several primary forest plots never show up in jungle rubber plots, even though the sample contains twice as many jungle rubber plots as primary forest plots. It is likely that the absence of those species, e.g. *Teratophyllum* spp. (Lomariopsidaceae) and *Trichomanes* spp. (Hymenophyllaceae), from the jungle rubber plots indicates that some primary forest species will never grow in jungle rubber.

6. Discussion and conclusions

6.1. Scale matters

The data clearly show that scaling relations differ between land use types and that plot level species richness does not directly indicate the (relative) richness of a land use type at the landscape scale. No single ratio can express the relative richness across different scales and conclusions as formulated by Leakey (1999) on the basis of the plot data of Michon and De Foresta (1995) cannot be trusted.

6.2. Conservation and production

Returning to the first question formulated in the introduction, we conclude that rubber production systems can indeed play some role in conservation of primary forest species (apparently providing forest-like habitat), but in places where primary forest is still present, priority should be given to conservation of remaining primary forest patches.

In places where primary forest is gone, jungle rubber can play a role in conservation of part of the primary forest species, while rubber plantations have little conservation value. In areas such as the Jambi lowlands where there is almost no primary forest left and where even logged-over forest is to a large extent already converted to plantations, jungle rubber might provide for intermediate levels of biodiversity while at the same time providing income to farmers (Van Noordwijk et al., 1997). ICRAF is currently working in this area on a project to increase income of smallholders and promote biodiversity conservation by keeping production in old jungle rubber on a profitable level. Techniques of gap replanting and direct grafting in rubber agroforests using genetically improved rubber are developed to extend the lifespan of existing rubber agroforests, at the same time reducing the frequency of slash-and-burn in the landscape (Wibawa et al., in review). With these techniques production could be raised while preserving the biodiversity associated with old jungle rubber.

6.3. Indicator groups

Species richness of terrestrial pteridophytes alone (without knowing the species or their ecological requirements) is not a useful indicator of habitat quality, as it discriminates poorly between the disturbed land use types and primary forest. A priori ecological information on the species involved is needed before terrestrial pteridophyte species can be used to indicate disturbance level or habitat quality of the forest understorey. If we would like to fully answer the question how much primary forest biodiversity is conserved in rubber agroforests we would have to sample most of the major taxonomic groups because different groups react in different ways to disturbance (see e.g. Thiollay, 1995 for birds). For each taxonomic group we would need enough samples to account for the variability in the data, and samples should cover a sufficiently large area to include different scales (plot level to landscape level). In addition, we need to know the ecological position (habitat requirements, guilds, etc.) of the species, as diversity alone does not give enough information for most taxonomic groups. Even so, such data collected within 'homogeneous' land use types cannot directly answer questions about the change in overall biodiversity value that can be expected if some types of land use will decrease, while others increase. The scaling rules within a land use type as given here will have to be (at least) complemented by assessments of species overlap between land use types. In addition assumptions have to be made about the maximum number of species present in each land use type as well as the minimum area required in each land use type to maintain healthy populations of those species.

It is understandable that available data are not compliant with all those requirements. Restricted by time and financial limits, researchers working in jungle rubber had to make choices with regard to the sampling dilemma, either researching all major groups but in small sample sizes and/or a small area, or getting ample information on one taxon and none on others. Difficult taxonomic groups in diverse tropical areas make the problem worse, as typically each sampling effort results in scores of new species to be named and described for the first time and existing ecological knowledge is limited. Pteridophytes proved in this study to be a relatively well-described group suitable to indicate local environmental conditions. Because the spores are wind dispersed their occurrence is not limited by presence of other organisms required for most seed dispersal or pollination. However, this characteristic of pteridophytes makes the group less suitable to represent biodiversity of other taxa. Hunting pressure and habitat fragmentation will affect some taxa more than others. Pteridophytes alone would probably provide us with a too optimistic view on biodiversity in jungle rubber.

As more results on different taxa become available it is no doubt possible to get a general idea of the order of magnitude of the contribution of jungle rubber to biodiversity conservation of tropical rain forest species. However, if the current trend of conversion to more intensively managed rubber or oil palm plantations continues we can be sure that hardly any biodiversity value will be left.

Acknowledgements

This research project was carried out in the framework of the Southeast Asian Regional Research Programme of the International Centre for Research in Agroforestry (ICRAF) in Bogor, Indonesia, as part of the 'Alternatives to Slash-and-Burn' Programme.

It was sponsored by UNESCO (1995–1997) and the Netherlands Organisation for Scientific Research (NWO, 1997–2000). We thank Hubert de Foresta for providing valuable advice at the start of the project, and Dr. P.H. Hovenkamp and Dr. H.P. Nooteboom of the National Herbarium of the Netherlands for checking species identifications.

References

Alston, A.H.G., 1937. The Selaginellae of the Malay Islands. II. Sumatra. Bull. Jard. Bot. Buitenzorg Série III 14 (2), 175–186.

Backer, C.A., Posthumus, O., 1939. Varenflora voor Java. Overzicht der op Java voorkomende varens en varenachtigen, hare verspreiding, oekologie en toepassingen. 's Lands Plantentuin, Buitenzorg, Java.

Beukema, H., Stolle, F., van Noordwijk, M., De Foresta, H., 1997. Biodiversity in rubber agroforests. Paper presented at the Smallholder Rubber Agroforestry Project Workshop in Bogor, Indonesia. ICRAF Southeast Asian Regional Research Programme, Bogor.

Colwell, R.K., 1997. EstimateS: Statistical Estimation of Species Richness and Shared Species from Samples, Version 5. User's Guide and application published at http://viceroy.eeb.uconn.edu/estimates.

Fletcher, W.W., Kirkwood, R.C., 1979. The Bracken Fern (*Pteridium aquilinum* L. (Kuhn)) its biology and control. In: Dyer, A.F. (Ed.), The Experimental Biology of Ferns. Academic Press, London, pp. 591–635.

Gouyon, A., De Foresta, H., Levang, P., 1993. Does 'jungle rubber' deserve its name? An analysis of rubber agroforestry systems in southeast Sumatra. Agrofor. Syst. 22, 181–206.

Holttum, R.E., 1932. On *Stenochlaena*, *Lomariopsis* and *Teratophyllum* in the Malayan region. Gard. Bull. (Singapore) 5, 245–313.

Holttum, R.E., 1938. The ecology of tropical pteridophytes. In: Verdoorn, F. (Ed.), Manual of Pteridology. Martinus Nijhoff, The Hague, pp. 420–450.

Holttum, R.E., 1959a. Gleicheniaceae. In: Van Steenis, C.G.G.J., Holttum, R.E. (Eds.), Flora Malesiana Ser. II Pteridophyta 1 (1). Martinus Nijhoff/Dr. W. Junk Publishers, The Hague, pp. 1–36.

Holttum, R.E., 1959b. Schizaeaceae. In: Van Steenis, C.G.G.J., Holttum, R.E. (Eds.), Flora Malesiana Ser. II Pteridophyta 1 (1). Martinus Nijhoff/Dr. W. Junk Publishers, The Hague, pp. 37–61.

Holttum, R.E., 1963. Cyatheaceae. In: Van Steenis, C.G.G.J., Holttum, R.E. (Eds.), Flora Malesiana Ser. II Pteridophyta 1 (2). Martinus Nijhoff/Dr. W. Junk Publishers, The Hague, pp. 65–176.

Holttum, R.E., 1966. Ferns of Malaya, 2nd ed. Botanic Gardens Singapore/Government Printing Office, Singapore.

Holttum, R.E., 1974. The fern-genus *Pleocnemia*. Kew Bull. 29 (2), 341–357.

Holttum, R.E., Hennipman, E., 1978. Lomariopsis group. In: Van Steenis, C.G.G.J., Holttum, R.E. (Eds.), Flora Malesiana Ser. II Pteridophyta 1(4). Martinus Nijhoff/Dr. W. Junk Publishers, The Hague, pp. 255–330.

Holttum, R.E., 1981. Thelypteridaceae. In: Van Steenis, C.G.G.J., Holttum, R.E. (Eds.), Flora Malesiana Ser. II Pteridophyta 1(5). Martinus Nijhoff/Dr. W. Junk Publishers, The Hague, pp. 331–560.

Holttum, R.E., 1991. Tectaria group. In: Nooteboom, H.P. (Ed.), Flora Malesiana Ser. II Pteridophyta 2(1). Rijksherbarium/Hortus Botanicus, Leiden.

Kramer, K.U., 1971. Lindsaea group. In: Van Steenis, C.G.G.J., Holttum, R.E. (Eds.), Flora Malesiana Ser. II Pteridophyta 1(3). Martinus Nijhoff/Dr. W. Junk Publishers, The Hague, pp. 177–254.

Kubitzki, K., 1990. The Families and Genera of Vascular Plants. Springer-Verlag, Berlin.

Laumonier, Y., 1997. The vegetation and physiography of Sumatra. Geobotany, vol. 22. Kluwer Academic Publishers, Dordrecht.

Leakey, R.R.B., 1999. Agroforestry for biodiversity in farming systems. In: Collins, W.W., Qualset, C.O. (Eds.), Biodiversity in Agroecosystems. CRC Press, Boca Raton, USA, pp. 127–145.

Michon, G., De Foresta, H., 1995. The Indonesian agro-forest model. Forest resource management and biodiversity conservation. In: Halladay, P., Gilmour, D.A. (Eds.), Conserving Biodiversity Outside Protected Areas. The Role of Traditional Agro-ecosystems. IUCN Forest Conservation Programme, Andalucía, pp. 90–106.

Page, C.N., 1976. The taxonomy and phytogeography of bracken—a review. Bot. J. Linn. Soc. 73, 1–34.

Pemberton, R.W., Ferriter, A.P., 1998. Old World Climbing Fern (*Lygodium microphyllum*), a dangerous invasive weed in Florida. Am. Fern J. 88 (4), 165–175.

Piggot, A.G., Piggot, C.J., 1988. Ferns of Malaysia in Colour. Tropical Press Sdn. Bhd., Kuala Lumpur.

Rosenzweig, M.L., 1995. Species Diversity in Space and Time. Cambridge University Press, Cambridge.

Spicer, R.A., Burnham, R.J., Grant, P., Glicken, H., 1985. *Pityrogramma calomelanos*, the primary, post-eruption colonizer of Volcán Chichonal, Chiapas, Mexico. Am. Fern J. 75 (1), 1–5.

Thiollay, J.-M., 1995. The role of traditional agroforests in the conservation of rain forest bird diversity in Sumatra. Conserv. Biol. 9 (2), 335–353.

Tomich, T.P., van Noordwijk, M., Budidarsono, S., Gillison, A., Kusumanto, T., Murdiyarso, D., Stolle, F., Fagi, A.M., 1998. Alternatives to Slash-and-Burn in Indonesia. Summary report and synthesis of phase II. ASB-Indonesia Report no. 8. ICRAF S.E. Asia, Bogor, Indonesia.

Van Noordwijk, M., Tomich, T.P., Winahyu, R., Murdiyarso, D., Suyanto, Partoharjono, S., Fagi, A.M., 1995. Alternatives to Slash-and-Burn in Indonesia. Summary report of phase I. ASB-Indonesia Report no. 4. ICRAF S.E. Asia, Bogor, Indonesia.

Van Noordwijk, M., Tomich, T.P., De Foresta, H., Michon, G., 1997. To segregate—or to integrate? The question of balance between production and biodiversity conservation in complex agroforestry systems. Agrofor. Today 9 (1), 6–9.

Wibawa, G., Hendratno, S., van Noordwijk, M., in review. Cyclical or permanent smallholder rubber agroforestry systems with or without slash and burn. In: Proceedings of the ASA Symposium on Alternatives to Slash and Burn.

Wong, K.M., 1982. Critical observations on Peninsular Malaysian Selaginella. Gard. Bull. (Singapore) 35 (2), 107–135.

Available online at www.sciencedirect.com

SCIENCE DIRECT®

Agriculture, Ecosystems and Environment 104 (2004) 113–134

Agriculture Ecosystems & Environment

www.elsevier.com/locate/agee

Biodiversity and ecosystem services in agricultural landscapes—are we asking the right questions?

M.J. Swift [a], A.-M.N. Izac [b], M. van Noordwijk [b,*]

[a] *Tropical Soil Biology and Fertility Institute of CIAT, Nairobi, Kenya*

[b] *International Centre for Research in Agroforestry (ICRAF), SE Asia Regional Program, P.O. Box 61, Bogor 16001, Indonesia*

Abstract

The assumed relationship between biodiversity or local richness and the persistence of 'ecosystem services' (that can sustain productivity on-site as well as off-site, e.g. through regulation of water flow and storage) in agricultural landscapes has generated considerable interest and a range of experimental approaches. The abstraction level aimed for, however, may be too high to yield meaningful results. Many of the experiments on which evidence in favour or otherwise are based are artificial and do not support the bold generalisations to other spatial and temporal scales that are often made. Future investigations should utilise co-evolved communities, be structured to investigate the distinct roles of clearly defined functional groups, separate the effects of between- and within-group diversity and be conducted over a range of stress and disturbance situations. An integral part of agricultural intensification at the plot level is the deliberate reduction of diversity. This does not necessarily result in impairment of ecosystem services of direct relevance to the land user unless the hypothesised diversity–function threshold is breached by elimination of a key functional group or species. Key functions may also be substituted with petro-chemical energy in order to achieve perceived efficiencies in the production of specific goods. This can result in the maintenance of ecosystem services of importance to agricultural production at levels of biodiversity below the assumed 'functional threshold'. However, it can also result in impairment of other services and under some conditions the de-linking of the diversity–function relationship. Avoidance of these effects or attempts to restore non-essential ecosystem services are only likely to be made by land users at the plot scale if direct economic benefit can be thereby achieved. At the plot and farm scales biodiversity is unlikely to be maintained for purposes other than those of direct use or '*utilitarian*' benefits and often at levels lower than those necessary for maintenance of many ecosystem services. The exceptions may be traditional systems where *intrinsic* values (social customs) continue to provide reasons for diversity maintenance. High levels of biodiversity in managed landscapes are more likely to be maintained for reasons of intrinsic, *serependic* ('option' or 'bequest') values or utilitarian ('direct use') than for *functional* or ecosystem service values. The major opportunity for both maintaining ecosystem services and biodiversity outside conservation areas lies in promoting diversity of land-use at the landscape and farm rather than field scale. This requires, however, an economic and policy climate that favours diversification in land uses and diversity among land users.

Keywords: Agricultural landscapes; Biodiversity; Ecosystem services; Functional groups; Resilience

1. Introduction

The role of biological diversity in the provision of ecosystem goods and services and the way this

* Corresponding author. Tel.: +62-251-315-234; fax: +62-251-315-567.

0167-8809/$ – see front matter © 2004 Published by Elsevier B.V.
doi:10.1016/j.agee.2004.01.013

role can be valued and managed during agricultural intensification is much debated but still poorly understood. A key problem in all debates on biological diversity is that the abstraction 'diversity' has often not been distinguished from the specific attributes of the community of organisms that is under study in any particular location or system. Likewise, evaluations of diversity have more often than not been assessments of the value of biological resources as such rather than assessments of the value of diversity per se (Nunes and van der Bergh, 2001). For instance, if the interest lies in the functional roles of the community these may depend on the 'structure' of the vegetation and the relationships between different 'functional groups', rather than on diversity as such (Woodward, 1993). Experiments based on random species assemblages may be appropriate tests for hypotheses about 'diversity' per se, but tell us very little about the largely self-selected assemblages that make up natural ecosystems. In the case of agroecosystems, whilst the dominant crops or livestock are human choices, by far the majority of the species (as soon as one takes the below-ground part of the system into consideration) are self-selected. So, are we asking the right question about the relations between biodiversity and ecosystem services? Does the loss of diversity at plot-to-global scales imply a threat to critical ecosystem functions? Can we identify thresholds in such a process?

Global diversity derives from the lack of overlap in species, genetic or agroecosystem composition between geographic or temporal domains. While 'agricultural development' directly affects local (i.e. plot level) diversity, it probably has even stronger effects by homogenizing at higher scales, facilitating the movement of 'invasive species' and the introduction and spread of 'superior' germplasm of desirable species. Scale is thus of overriding importance in our analysis and we may well find that answers may appear contradictory between different ways of defining temporal and spatial boundaries to the system under consideration. In this review we will first consider the concepts of 'biodiversity' and 'ecosystem functions', and then the evidence that links relevant aspects of the two, before we embark on an exploration of how this relationship depends on scale and can be 'managed'.

2. The biological basis of ecosystem goods and services

Humans have evolved as part of the world's ecosystems, depending on them for food and other products and for a range of functions that support our existence. Natural ecosystems, as well as those modified by humans, provide many services and goods that are essential for humankind (Matson et al., 1997). Efforts and interventions to manipulate (agro)ecosystems in order to meet specific production functions represent costs to the rest of the ecosystem in terms of energy, matter and biological diversity, and often negatively affect goods and services that so far were considered to be free and abundant. These are anthropocentrically regarded as services because they provide the biophysical necessities for human life or otherwise contribute to human welfare (UNEP, 1995; Costanza et al., 1997). Most if not all of these services are based on a 'lateral flow', or movement across the landscape of biomass (such as food, fibre and medicinal products derived from the sea, inland waters or lands outside of the domesticated 'agricultural' domain), living organisms and their genes, or earth (nutrients), water, fire or air elements. Examples of ecosystem services particularly important for agroecosystems and agricultural landscapes are: maintenance of the genetic diversity essential for successful crop and animal breeding; nutrient cycles; biological control of pests and diseases; erosion control and sediment retention; and water regulation. At a global scale other services become important such as the regulation of the gaseous composition of the atmosphere and thence of the climate. A list of such services is given in the first column of Table 1 and Appendix A, and their connection to lateral flows is discussed by Van Noordwijk et al. (this volume, Table 1).

These ecosystem goods and services are biologically generated. The community of living organisms within any given ecosystem carries out a very diverse range of biochemical and biophysical processes that can also affect neighbouring systems. These can be described at scales ranging from the subcellular through the whole organism and species populations to the aggregative effect of these at the level of the ecosystem (Schulze and Mooney, 1993). All ecosystems have permeable boundaries with respect to material exchanges but the within-system flows usually dominate those between systems, such as between land-use or land-cover types within a landscape. For purpose of this paper we define *ecosystem functions as the minimum aggregated set of processes* (*including biochemical, biophysical and biological ones*) *that ensure the biological productivity, organisational integrity and perpetuation of the ecosystem*. There are no agreed criteria for defining a minimum set of such functions but for the purposes of this paper the second column of Table 1 lists ecosystem functions alongside the ecosystem services they provide. Further explanation of these relationships is given below but it is useful to note that these functions can be pictured as having a hierarchical relationship. The energy captured in primary production is utilised in the herbivore and decomposer food chains. Interactions between these three subsystems occur through nutrient exchanges and a variety of biotic regulatory mechanisms as well as by energy flow. In particular, the balance between the constituent processes of primary production and those of decomposition determines the amount of energy and carbon maintained within the system and is the major natural regulator of the gaseous composition of the atmosphere at a global scale (Swift, 1999).

3. Biological diversity and its values

Most discussions and empirical studies on biodiversity have focused on issues of a relatively small range of organisms. In contrast, the Convention on Biological Diversity defines its area of concern as:

"... the variability among living organisms from all sources, including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species, and of ecosystems" (Heywood and Bates, 1995). Diversity within each one of these three fundamental and hierarchically related levels of biological organisation can be further elaborated as follows: genetic diversity is the variation within and between species populations; species diversity refers to species richness, that is, the number of species in a site, habitat, ecological zone or at global scale; ecosystem diversity means the diversity of assemblages (and their environments) over a defined landscape, ecological zone or at global scale.

Biodiversity in this paper refers to the totality of the species (including the genetic variation represented in the species populations) across the full range of terrestrial organisms, i.e. invertebrate animals, protists, bacteria and fungi, above- and below-ground, as well as the vertebrates and plants which often constitute the main concerns of biodiversity conservation. With a definition as broad and inclusive as this, it is highly unlikely that any clear and precise statements about relationships between 'biodiversity' and functions can be formulated and tested that can be helpful in guiding human activity. Similar to the situation with 'watershed functions', which are considered in the next part of this volume, we may find that discussions on components of the overall biodiversity concept in relation to land-use are more productive and open to progress than those that stay at the aggregate level. In the section immediately following we shall refer to the diversity within ecosystems (often termed alpha diversity) and in later sections to that at the broader scale of the landscape (which embraces concepts of both beta and gamma diversity).

The analysis of biodiversity and its management are highly influenced by the perspective used. In particular, different sectors of society attribute different values to biodiversity. Since biological diversity concerns different levels, from genes to species and ecosystems, the value of diversity can likewise be defined in a number of different ways. Broadly speaking, four different types of value can be usefully recognised, although different terminology is often used by different authors (see Nunes and van der Bergh, 2001 for further details).

First is the *intrinsic* (sometimes called 'non-use') value of diversity to humans, or the value that biodiversity has on its own. This value comprises cultural, social, aesthetic, and ethical benefits. Some groups in society attribute high social and religious values to individual species or communities of organisms; others derive value from the simple fact of high diversity per se in such systems as tropical rainforests or coral reefs.

Second is the *utilitarian* (also called direct use, contributory, primary or infrastructure) value of components of biodiversity. These are the subsistence and commercial benefits of species or their genes derived by one or other sectors in society. The utilitarian value may be private and accrue to the land managers (farmers, local community, government). This is most obvious with respect to high value agricultural crops but also applies to the other types of good listed in Table 1. Utilitarian value may also accrue to other sectors in society, in addition to private land managers. For instance, the pharmaceutical industry values the tropical forest tree *Prunus africana* very highly because its bark contains chemicals used for manufacturing a drug. Another example is that in Africa, many farmers living near natural (and protected) forests withdraw substantial monetary benefits from their hunting and from collecting plants and tree products in these forests (Pottinger and Burley, 1992). Utilitarian value thus refers to the use of organisms that are part of the local diversity as inputs into consumption and production processes.

Thirdly, biodiversity can be said to have *serependic* (‘option’, or bequest) value. This is the belief in future but yet unknown value of biodiversity to future generations, for example, the presence of a microorganism with an as-yet undiscovered genetic potential for industrial products. These three types of value of biodiversity are ethnocentric and depend very much upon the cultural values and preferences of different sectors of society. This is why some authors, interested in such values, stress that ‘the conservation of biological diversity depends as much on society’s ethical views as on facts’ (Barrett, 1993).

Finally, biodiversity contributes to ecosystem life support functions and the preservation of ecological structure and integrity. We refer to these functions as the *functional* value of diversity. This category of value has only been relatively recently recognised in the economic literature as an important category per se which overlaps partially with concepts such as that of ‘indirect use’ value (see Kerry-Turner, 1999). Part of this functional significance may result in direct utilitarian value for *Homo sapiens* in the production of goods and services that can be priced. Beyond this lie a range of ecosystem services that are of acknowledged benefit to humans but which generally lie outside the boundaries of recognised direct utilitarian benefit. The purpose of this paper is to analyse the functional values of biodiversity with particular reference to the diversity in agricultural landscapes.

4. What is the relationship between diversity and function?

4.1. Concepts

Biologists have for many decades speculated on the question of why there are so many species of living organisms. As explored in the theory of island biogeography, the diversity within any ecosystem at any point in time is the result of a ‘self-selection’ process that involves co-evolution of the species comprising the biological community within a given ecosystem by interactions among them and with the abiotic environment through time. This is not an isolated process. New species may enter an ecosystem from neighbouring areas, some establishing themselves and others failing to do so. Partly as a result of successful newcomers or new adaptations emerging in existing ones (be they competitors, predators, pests or diseases), and partly as a result of fluctuations in abiotic environmental conditions, some of the existing species may become (locally) extinct over any period of time. The species richness of any given ecosystem or land unit is therefore a dynamic property. Recently, the conventional explanation of local diversity as well as its ‘functionality’ embodied in the niche concept has been challenged by theories that derive patterns close to the observed ones from ‘random walks’ in abundance of species without any a priori prediction of the direction of selection pressures and based on an equivalence of intra- and interspecific competition (Hubbell, 2001).

In agroecosystems farmers take a dominant role in this dynamic by the selection of which organisms are present, by modifying the abiotic environment and by interventions aimed at regulating the populations of specific organisms (‘weeds’, ‘pests’, ‘diseases’ and their vectors, alternate hosts and antagonists). The dynamic nature of the (local, patch level) diversity of any system, whether natural or agricultural, is often underrated, as is the importance of the selection pressure and process. The diversity of any system is not adequately represented simply by the number of species (or genotypes) present, but by the relationships between them in space and time. Attempts to assemble combinations of the same number of species under slightly different conditions and in particular without the history of interaction often fail (Ewel, 1986, 1999). But what makes any existing species combination into a 'system' is still largely elusive. Some insights obtained in analysing food webs may help. For example, Neutel (2001) showed that the majority of below-ground food webs constructed from random combinations of organisms did not meet dynamic stability criteria, even though all parameters such as abundance of groups and dynamic properties were chosen in a 'normal' range when considered one-by-one. Yet, systems with the actual parameter combinations that are attained in the field did meet stability criteria, suggesting that partly uncovered rules about the proportionalities and co-variance within the normal range are crucial.

Debate on the relationship between biological diversity and ecosystem function has a long history which has taken on new vigour (and sometimes even rancour) since the advent of the Convention on Biological Diversity (see Woodwell and Smith (1969) for the older literature and Schulze and Mooney (1993), Mooney et al. (1995, 1996), and many of the citations below for more recent discussion). Vitousek and Hooper (1993) contributed a major focus to this debate through hypothesising three different possible relationships between plant diversity and broad-based ecosystem functions such as the rate of primary production (Fig. 1). Their analysis of current evidence led them to propose that the asymptotic relationship shown as Curve 2 in Fig. 1 was the correct one. This suggests that whilst the essential functions of an ecosystem, such as primary production, require a minimal level of diversity to maximise efficiency this effect is saturated at a relatively low number. Swift and Anderson (1993) proposed that this relationship could also apply to the decomposer system. Examples of essential functions in this case are the basic suite of catabolic enzymes (e.g. for cellulolysis, lignin degradation, etc.), the facilitation role that invertebrates play by reducing particle size by their feeding activity, and biophysical processes of pore formation and particle aggregation. It is interesting to note, however, that the communities of organisms contributing to the ecosystem function of decomposition are taxonomically much more diverse than those of primary production.

4.2. Experimental approaches

Over recent years a number of authors have reported on experiments investigating the links between diversity and specific functions (e.g. see Ewel et al., 1991; Naeem et al., 1994; Naeem and Li, 1997; Tilman and Downing, 1994; Tilman et al., 1996, 1997; Hooper and Vitousek, 1997) that appear to broadly corroborate the predictions of the Vitousek–Hooper hypothesis for primary production. This has however generated an equal amount of discussion in refutation and the issue remains significantly a matter of interpretation and opinion (see Grime, 1997; Hodgson et al., 1998; Lawton et al., 1998; Wardle et al., 2000; Naeem, 2000). There is no space here to review these studies in detail, but refer to Kinzig and Pacala (2001) and

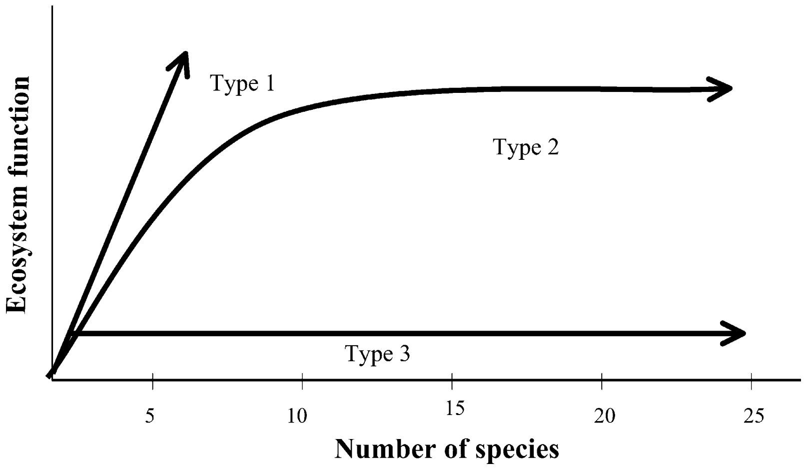

Fig. 1. Possible relationships between biological diversity and ecosystem functions for the plant subsystem (from Vitousek and Hooper (1993)). The authors hypothesised that Curve 2 was the most probable of the three propositions.

Tilman and Lehman (2001) for a synthesis that acknowledges the 'sampling effect' that probably dominates the initial phases of the experiments and the fact that 'niche assembly' will be a relatively slow process, especially where we are interested in stages beyond the pioneer phase. Each one of the experiments quoted can be criticised in one way or another. The strictest interpretation of many of the experiments would be that the conclusions apply only to the specific combinations of organisms used in the tests, and in most cases these are assemblages constructed for experimental purposes rather than naturally co-evolved communities. At a fundamental level such experiments suffer from a basic methodological paradox—in order to describe and understand diversity and complexity we need to simplify it, and take away the self-selection that governs real-world diversity. Dealing with the totality is impossible. For instance, there is no single (or combination of) methods that would allow for the total inventory of the species richness of even a small volume of soil. It is thus difficult to draw general conclusions about 'diversity' as such and in particular with respect to naturally co-evolved communities. The results of such 'un-natural' experiments may, however, be more applicable to agricultural systems that in one sense can be said to have been assembled in a similar way.

4.3. The minimum diversity required within a functional group

One potentially valuable interpretation of the Vitousek–Hooper relationship has been that the minimal level of diversity required to maximise the production function consists of representatives of an essential set of 'functional groups' of plants (Schultze and Chapin, 1987). A functional group may be defined 'a set of species that have similar effects on a specific ecosystem-level biogeochemical process'. As Vitousek and Hooper put it, the 'essential' plant species are those that contribute in different ways to the key ecosystem functions—in the case of primary production by exploiting different components of the available resources by differences in canopy structure to maximise light capture or symbionts and root architecture to optimise capture of water and nutrients. Drawing together the threads of this discussion we hypothesise that 'the minimum diversity essential to maintain any given ecosystem function can be represented by one or a few functionally distinct species i.e. one or a few representatives of a small range of functional groups' is a useful null-hypothesis to guide investigations of the functional significance of biological diversity in agricultural systems. It may need further operationalisation for specific ecosystem contexts, however. The total diversity required then depends on the number of functions that are recognised and to the degree of overlap in 'functional groups' between these different functions.

5. Which functional groups of organisms are essential?

The functional group concept is briefly discussed in Appendix A to this paper and Table 1 lists a minimal set that we propose are needed to provide the ecosystem goods and services we have been addressing.

The classification of plants into functional groups has drawn a great deal of recent attention because of the recognition of the pressure being exerted on terrestrial ecosystems by global climate change (Smith et al., 1997). The primary producers (together with the vertebrate herbivores) are our major source of food and are also the source of fibre and other useful materials such as latex. Molecules with antibiotic, therapeutic, pesticidal or similar biological activities utilised by humans are, however, synthesised by many groups of organisms (e.g. bacteria and fungi) and are often very specific in origin. Diversity is therefore an essential pre-requisite for maintenance of supply, particularly of new products, although the capacity to biologically generate or synthesise new compounds under laboratory conditions has been greatly increased by the advent of genetic engineering.

Decomposition and mineralisation of organic matter of plant and animal origin and synthesis and decomposition of soil organic matter are carried out by a very diverse community of invertebrates, protists, bacteria and fungi. Other elemental transformations often are carried out by a diverse set of functional groups with very specific biochemical capacities, for example, certain bacteria of the nitrogen cycle. Diversity within these groups varies from very low to high, but it can be experimentally demonstrated that a single species per function may be sufficient under a given set of environmental conditions.

The dominant biological properties regulating water flow and storage in the soil are the plant cover, the soil organic matter content and soil biological activity. Macrofauna such as earthworms, termites and other invertebrates influence the pore structure. Bacteria and fungi modify the extent of aggregation of soil particles. All these organisms and an additional range of decomposer organisms influence synthesis and decomposition of soil organic matter. Control of erosion and trapping of sediment is regulated by the architecture of the plants at and below the soil surface, the amount (and hence the rate of decomposition and movement) of surface litter, and the physical quality and organic matter content of the soil.

Under natural conditions the interactions between the populations of organisms at the various trophic levels i.e. plants, herbivores, symbionts, parasites, decomposers, predators and secondary predators result in a dynamic balance of population sizes. The total diversity is huge but any single population is only influenced by a relatively small number of interactions. Biological regulation of a specific pest, pathogen or disease vector of interest to humans is therefore dependent on a significant level of diversity among its parasites or predators. These in their turn may depend on other elements of diversity for their survival, e.g. the presence of microhabitats, alternative hosts, nesting or egg laying sites, or refuges often provided by the vegetation.

Chemical transformation of toxic organic elements, chelation or absorption of basic elements and removal of toxic levels of nutrients or other chemicals from ground, running or soil water may be carried out by a diverse range of bacteria, fungi or protists often in association with invertebrates. In well-established waste disposal systems these organisms form ‘guilds’ which function in a very integrated way. As with decomposers distinct guilds may operate across different ranges of environmental gradients of temperature, pH, moisture, etc.

The earth’s climate is regulated by the content of ‘greenhouse’ gases in the atmosphere (CO_2, CH_4, NO_x, etc.). Carbon dioxide is emitted or taken up under one circumstance or other by the majority of living organisms and is thus a phenomenon of such generality as to defy attempts to relate its dynamics to changes in diversity other than the totally catastrophic. Methane and the nitrous oxides are, however, the product and/or substrate for a relatively small number of bacterial species in the soil associated with soil, decomposing organic matter or the gut flora of animals. Diversity change may thus be more significant in these cases.

It is worth noting that even when the discussion of function–diversity relationships is reduced to considering only functional groups, the minimum extent of necessary diversity that is implicated is still very high.

5.1. *What is the significance of diversity within functional groups?*

If the above hypothesis is correct and ecosystem functions can be maintained by the minimal number of representatives of the essential functional groups, then the questions remains as to what is the significance of the often high diversity within functional groups—which takes us back to the basic biodiversity question ‘why are there so many species’? Answers to this question depend strongly on the scale of consideration. Different species often occupy similar ecological roles in geographically separated areas, and one of the major threats to local species is the lateral flow of organisms once such geographical barriers disappear. Replacement of local species by intrusive exotics does not necessarily change ecosystem processes, or local richness, although there are dramatic exceptions for specifically successful (from the perspective of the invader, at least) invasions. Such invasions are likely, however, to reduce global diversity and in fact have been identified as one of the major drivers of ‘global change’.

Vandermeer et al. (1998) summarised the main issues in the discussion on the role of diversity in agroecosystems in the following three hypotheses of links between diversity and function:

1. Biodiversity enhances ecosystem function because different species or genotypes perform slightly different functions (have different niches);
2. Biodiversity is neutral or negative in that there are many more species than there are functions and thus redundancy is built into the system;
3. Biodiversity enhances ecosystem function because those components that appear redundant at one point in time become important when some environmental change occurs.

It is valuable to note that these are not necessarily mutually exclusive hypotheses, as they may refer to different space and/or time aspects of the system and the function of specific concern. We need to clearly separate the question of how the current diversity came into being (the ‘self-organisation’ of the system, based on the success in the evolutionary history of all component species) from the human or teleological perspective of the relevance of this diversity. Just as we have to distinguish between ‘diversity per se’ and ‘diversity of actual systems’, we have also to recognise that not all components of a system have the same probability of being lost as a result of simplification of agroecosystems and some functions may therefore be more resilient than others. Differences in life histories of the key groups of organisms confer different temporal and spatial contexts to their role in the ecosystem and their responsiveness to its self-organising properties.

The third of Vandermeer et al.’s (1998) hypotheses is extremely pertinent to the question of how much of this diversity is needed to maintain ecosystem goods and services in the face of agricultural intensification and other aspects of ongoing ‘global change’. There is certainly substantial experimental evidence that the many key functions can be maintained by only small numbers of species within a particular functional group. For example, monotypic cover by perennial plants can be as effective as a diverse community in controlling erosion. Although the decomposer community of a particular soil may be very diverse, only a minority of the hundreds of species of fungi, bacteria or invertebrates participate in the decomposition process at a given time and place. The extent of redundancy implied by this can be demonstrated under laboratory conditions where decomposition can be fully mediated by single species cultures of enzymatically diverse organisms such as white-rot basidiomycete fungi whilst in nature the same process may be carried out by several species of fungi, bacteria and animals (Swift, 1976; Giller et al., 1997).

The third hypothesis raises questions whether key functions can be maintained by one (and the same) species under all circumstances. This addresses the issue of the capacity of ecosystems to adapt to changing circumstances that result from elements of stress and disturbance. The capacity of a system to respond to and recover from disturbance is termed its resilience. This property has been attributed to the degree of connectivity within an ecosystem, a feature that depends at least in part on the composition and diversity (Holling, 1973, 1986; Allen and Starr, 1982). Diversity within functional groups may provide an important means for increasing the probability that ecosystem performance can be maintained or regained in the face of changing conditions. For the below-ground community, for instance, there is evidence that the same enzymatic function is carried out by different species of bacteria or fungi from the same soil under different, and even fluctuating, conditions of moisture stress or pH (see Griffin (1972) for discussion of this). In the case of plants different species may play a similar functional role in different seasons, under varying conditions of climatic or edaphic stress and in different stages of patch-level succession.

6. Resilience and diversity thresholds

Functional diversity thresholds are thus likely to be higher in the real world than in the relatively controlled situations under which most of the experiments on diversity–function relationships have been conducted. Recognition of the importance of diversity to the property of resilience suggests furthermore that the implication of equilibrium in the way that Fig. 1 is drawn (see also Figs. 2 and 3) may be misleading. The shifts between different states of functional efficiency with changes in diversity are more likely to be rather abrupt. Perhaps a case could be made recognising resilience

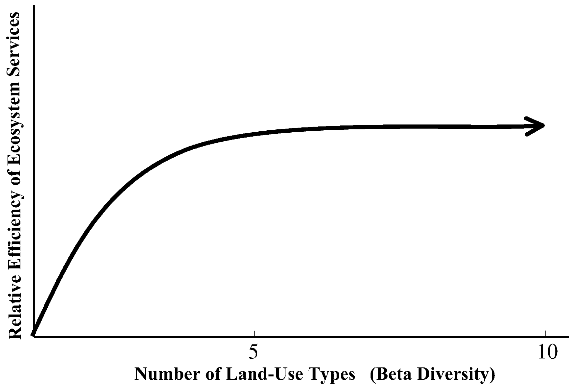

Fig. 2. Hypothesised relationship between the diversity of ecosystem or land-use types and the efficiency of function of (the totality of) ecosystem services at the landscape scale.

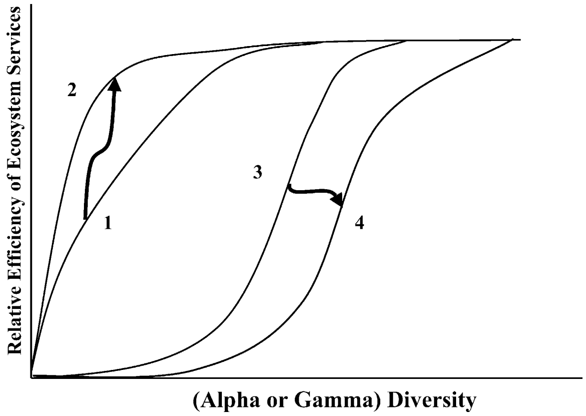

Fig. 3. Hypothesised relationships between diversity (as measured by species richness) and the efficiency of function of ecosystem services at the patch-ecosystem (i.e. plot) scale (Curves 1 and 2) and the scale of the landscape (Curves 3 and 4). Curve 1 repeats hypothesis 2 of Fig. 1: Curve 2 shows how in an intensively managed agricultural plot ecosystem services may be maintained by substitution of diversity by inputs derived from human and petro-chemical energy. Curve 3 shows, by comparison with Curve 1, that the threshold of 'essential' diversity is greater as the land area increases. Curve 4 represents circumstances of high disturbance of the landscape by human intervention.

as an ecosystem service rather than a property. An alternative view, however, is to see resilience as a property which varies among functions rather than a unitary ecosystem property. The decomposition function, for example, may be substantially more resilient than that of the regulation of specific pest populations.

Resilience is a concept that requires consideration at different spatial scales. The resilience of any local system after shocks that lead to local loss of diversity depends strongly on the ability of organisms to recolonise from the neighbourhood, and thus on the distance to the nearest suitable habitat and the dispersal of the organisms in question.

7. Managing biodiversity and ecosystem services in agricultural landscapes

7.1. What is the impact of agricultural intensification on biodiversity and ecosystem functions?

Our main concern in this paper is with biodiversity issues in agricultural landscapes, i.e. landscapes containing agroecosystems. Agroecosystems can be defined as (natural) ecosystems that have been deliberately simplified by people for purpose of the production of specific goods of value to humans. The simplification down to one or a few productive plant or animal species is implemented for greater ease of management and specialisation of product to suit market demands, especially in highly mechanised forms of agriculture. In an ecological sense the system may be seen as one which is maintained by a high frequency of disturbance, in an early successional stage (Conway, 1993). In such systems a distinction has been made between 'planned' and 'associated' diversity (Swift et al., 1996; GCTE, 1997). The *planned diversity* is the suite of plants and livestock deliberately retained, imported and managed by the farmer. The composition and diversity of this component strongly influences the nature of the *associated biota*—plant, animal and microbial. The issue is more complex than the single issue of the extent of planned biodiversity that is maintained, however. Agroecosystems are managed by substitution and supplementation of many of the natural ecosystem functions by human labour and/or by petro-chemical energy or its products.

In addition to their direct effects on production these interventions provide the means to reduce the risk associated with reliance on ecosystem services, although it can be argued that this is serving to substitute one set of risks for another—that of dependence on the market. Furthermore, whilst substitutions may buffer some of the functions they also run the risk of further damaging others. For instance, the addition of pesticides may control diseases of immediate negative impact but also kill non-target organisms with other functions such as pollination or soil fertility enhancement.

During agricultural intensification the diversity of crops and livestock is reduced to one or a very few species of usually genetically homogenous species. The varieties are selected or bred for yield (e.g. high plant harvest index), taste and nutritional quality. Plant arrangement is commonly in rows, fallow periods are bare, sequences may be monospecific (varietal) or of two or rarely more species. This is in contrast to natural ecosystems where the genetic diversity of plants (both within and among functional groups) is high but varies in relation to environment. The effects of land-use change and agricultural intensification on biodiversity and associated functions are still poorly understood but conversion to agriculture almost always results in fewer species of both planned and associated biota with lower genetic variation and representing less functional groups. Nonetheless the extent of diversity in even so-called monocultures may be underestimated by plot-level assessment of diversity at any point in time. A rapid interannual turnover of the germplasm is often employed to stay ahead of the evolutionary race with pests and diseases, adding a time dimension to diversity that may exceed evolution in natural systems, albeit with respect to a narrow genetic base. This varietal turnover depends, however, on 'externalised' functions of maintaining genetic diversity in gene banks, and on the mechanisms of rapid multiplication and transfer of such germplasm. This situation contrasts with that of extensive agricultural systems where diversity is deliberately maintained within the system with or without external exchange. Here a plot-level assessment may have more relevant boundaries of measurement, although lateral flows of organisms exist here as well. Production systems based on perennial crops and trees provide less opportunity for rapid turnover of varieties for obvious reasons, and there clearly is a much stronger need here for maintaining plot-level diversity as a risk management strategy (Van Noordwijk and Ong, 1999).

7.2. Primary production

Whilst many recent experiments have tended to confirm that community primary production may be maximised by a low-number diversity of functional types (see above) there is also abundant evidence that monotypic stands can reach the same levels of production within relatively narrow environmental conditions. Biomass production is, however, not the only function or service performed by plants in ecosystems. The secondary functions related to ecosystem services may be more biodiversity-sensitive than that of food production. 'Intensive' production systems for specific high-value products (e.g. spices) can, however, be very diverse. Another exception may be in relation to pharmaceutical and agro-chemical goods. Most products of these types are initially gathered from natural or secondary vegetation or derived from microbial cultures obtained from soil. Once the markets for such products are established, however, the required control over the concentrations of biologically active substances tend to favour more technically advanced and intensive modes of production. Maintaining global diversity is thus essential for both present and future needs although the synthetic capacity brought by the molecular biological revolution is fast rendering this less so. Herbivore diversity is highest in heterogeneous systems with high plant and resource diversity but monotypic vertebrate herds can reach equivalent levels of production in simplified grazing systems. Pest epidemics tend to occur in circumstances of low genetic diversity of the host plants or livestock.

7.3. Nutrient cycling

Nutrient cycles become more open in agricultural systems with losses of nutrient through offtake in harvest, run-off from compact surfaces, increased volatilisation through a changed surface environment and increased leaching associated with decreased soil organic matter content. These losses can be substituted by inorganic inputs but the efficiency of return to the plant is often low and fertilisation is usually required at levels far in excess of direct crop demand, which further exacerbates the losses and can leads to pollution of groundwater, etc. There is substantial evidence demonstrating gains in crop productivity from nutrient additions through mixtures of organic and inorganic sources of nutrients compared with either alone (e.g. Swift et al., 1994). Maintenance of organic inputs to the soil is thus an important management strategy for efficient use of external inputs. Advantages in utilising a variety of such inputs have also been demonstrated because of the strong influence of input chemistry ('resource quality') on patterns of mineralisation. The diversity of organisms involved in nutrient cycling may be substantially reduced under agricultural intensification but there is little evidence of significant effects on decomposition and mineralisation processes which has been attributed to a high level of functional redundancy among decomposer fungi, bacteria and microregulators such as nematodes or collembola (Beare et al., 1994, 1997; Giller et al., 1997). The significance of this loss of diversity should not, however, be assumed to be inconsequential. In particular, it is unclear how the resilience of the system under conditions of change is influenced by such loss. Organisms with very specific functions, such as those exhibited by some bacteria of the nitrogen cycle, often show specialisation to particular soil conditions such as pH and specific genotypes may be lost as a result of soil degradation. Specific strains of di-nitrogen-fixing bacteria may also be lost as a result of agricultural intensification resulting in the need for subsequent inoculation (Kahindi et al., 1997).

7.4. Organic matter dynamics

Soil organic matter (SOM) is a keystone component of the ecosystem in the sense that its impact on overall system performance exceeds its relative share in the energy flow through the system. Soil organic matter stores and buffers nutrient concentrations, influences water storage in the soil and is a major factor in determining soil structure and thence erosivity. Above all, it is a store of energy in the soil that drives many of the soil-based processes. SOM synthesis and decomposition is brought about by much the same community of organisms as those involved in decomposition of plant litter. A well-charted phenomenon is the decline in SOM as a result of conversion of natural ecosystems to agriculture. Farmers utilise the nutrients mineralised as part of this decline of the SOM capital to support high initial levels of crop production after clearance. Soil tillage is also an effective additional way of stimulating the breakdown of SOM and plays a key role in promoting crop yields after land conversion to agriculture, until a new and lower equilibrium between breakdown and formation of SOM is reached. The level of the new SOM equilibrium, with its consequent impact on nutrient cycling, soil water regimes and erosivity, is related to the quantity of plant litter input, which is almost invariably lower than that of natural systems. Crops in intensive systems are usually selected for high harvest indices, and there may be uses for crop residues other than soil fertility maintenance (e.g. fodder or fuel). The SOM content is thus related to the quantity, diversity and mode of management of organic input to soil. A key feature of agroecosystem management is thus the trade-off between the gains in production from 'mining' the SOM versus the potential negative impact on its other ecosystem services and in particular on system resilience. This 'trade-off' between the different values of SOM has been rarely recognised but became a matter of greater interest as society has begun to realise the potential value of sequestering carbon in soil as a means to slow down the rate of global climate change. A research question of continuing interest is whether the functional properties of SOM are in any way influenced by the diversity of organic materials from which it is synthesised.

7.5. Watershed functions

The most important factors regulating water infiltration and retention are the extent of ground cover by plants and/or plant litter. The reduction in these, including interposing of periods when ground is bare, leads to greater run-off and diminished infiltration as well as increasing the risk of erosion. Substitution by mechanical tillage can ameliorate as well as aggravate these effects. Monospecific cover can be just as effective as a diverse one with respect to limiting run-off and erosion, trapping sediment and promoting infiltration, but to be effective it has to be present year round. Diversity of organic inputs is likely to have a positive effect by widening the probability of differences in timing of litterfall and rates of disappearance from the soil surface. As soil protection on slopes depends more on partially decomposed litter with good ground contact than on fresh leaves that can be easily washed away, the role of plant diversity on slopes is likely to be greater than on flat lands. The macrofauna moving between litter layer and soil strongly influence partitioning of water between surface runoff and infiltration as well as modifying water movement within soil. Interesting examples of the influence of these 'ecosystem engineers' show how circumstance-specific diversity effects may be. Soil engineers making macropores in the soil are not welcome in all circumstances. In bunded rice fields, farmers make an effort to destroy soil structure by puddling to reduce the porosity of the soil and building dykes to contain the water. These earthworks may be destroyed by the actions of earthworms and surveys by Joshi et al. (1999) in the Ifugao Rice Terraces (IRT), in the Philippines showed that 125 out of 150 farmers interviewed ranked earthworms as the most destructive pest of terraced rice fields. In a second example, the conversion of Amazonian rainforest to pastures has been shown to lead to extinction of the natural earthworm community, which have been replaced in some circumstances by a single exotic species, *Pontoscolex corethrurus*. This has a negative effect on pasture productivity because the introduced worms compact the soil, whereas the native species improve soil structure (Chauvel et al., 1999). Inoculation with species from the forest might reverse this effect, but remains to be tested.

7.6. Risks of pests and diseases

As already indicated the decreased genetic diversity of plant cover increases the risk of pest attack. Simplification of the ecosystem and in particular the use of broad-spectrum pesticides also decreases the diversity of natural enemies and increase risks of pest attack (Lawton and Brown, 1993). Pesticides also have negative effects on non-target beneficial organisms including pollinators and beneficial soil biota.

7.7. Greenhouse gas emissions

Land-use change alters the balance of gas emissions and thence influences global climates. There are very large increases in the CO_2 output during clearing from natural vegetation and breakdown of soil organic matter reserves that are rarely if ever balanced by regrowth. The output of methane may be significantly increased in systems such as paddy rice and intensive cattle production and of nitrous oxides by N-fertilisation. These changes are linked to alterations in soil structure that dominate changes in the activity of a variety of soil organisms (e.g. methanogenic and methanotrophic bacteria) but we are not aware of any documented case where such effects are linked to the absence of functional groups or to biodiversity change per se.

7.8. A hierarchy of functions

There are a few general conclusions that may be drawn from this brief review of the impacts of agricultural intensification on the relationship between biodiversity and ecosystem services. *First*, that whilst there are a number of clear examples where changes in diversity have threatened the provision of ecosystem services, especially relating to the regulation of pests and diseases, there are also others where the changes in biodiversity seem to be functionally neutral, at least within relatively stable environmental conditions. *Second*, there may be some functional groups, particularly microorganisms such as the decomposers, where the degree of functional redundancy is such that the resilience of the function is very high. These two observations may be generalised by stating that there are no rules to be derived for agricultural systems concerning the importance of biodiversity with respect to the maintenance of ecosystem services that apply across all functional groups and environmental circumstances. Both the concept of 'diversity' and that of 'ecosystem function' are too broad to make generalisations at this level testable. There is a need and potential, however, to investigate the issues of thresholds of diversity–function relationship within specific functional groups and under circumstances of change in stress and/or disturbance.

Finally we should re-emphasise the importance of the hierarchical control exerted by the plants over the other functional groups (Fig. 4, Appendix A).

This is a particularly important feature when determining management options, not only at the field and farm scale but also at that of the landscape. The plant, decomposer and herbivore subsystems of the biological community interact in a variety of ways but the productivity, mass, chemical diversity (resource quality) and physical complexity of the plant component exerts the strongest influence and is the single most important determinant of both the diversity and the functional efficiency of the other two subsystems. Wardle et al. (1999a,b) and Yeates et al. (1999) showed, for example, that arthropod and microbial communities were not adversely affected by agricultural intensification provided the type of management (e.g. mulching) provided for increases in the quantity and quality of the organic inputs. The maintenance of total system diversity and of the major part of the ecosystem services is thus predominantly determined by the nature of the plant community. This is also of course the main point at which humans intervene in the agroecosystem—to decide the species richness, the genetic variability and the organisation in space and time of the planned biota in the vegetation subsystem.

8. Implications for the design and management of agricultural landscapes

A substantial research investment has been made into agricultural systems that fall short of the full extent of genetic homogenisation and petro-chemical substitution. Examples are agroforestry and other inter-crops, rotations, mulch-based, minimum tillage and integrated livestock-arable systems. All these systems are characterised by maintenance of diversity of plant functional groups above the level of monocropping. The scientific justification for such approaches has generally been made on grounds of greater functional sustainability and the wider spread of risk associated with more diverse products as well as on the recognition that it is in line with the management choices of the majority of the rural poor in the tropics. For farmers labour saving and low investment and risk may be the preferred attributes of these systems. It is interesting to note however that, whereas scientists had introduced single-species fallow systems to farmers in Western Kenya, these farmers decided on their own to diversify tree species in these improved fallow systems (Bashir Jama, pers. commun., 2001).

The simplicity of monocultures at field level is only possible as long as farms are part of a germplasm delivery system with rapid access to externalised gene banks and have access to risk buffering mechanisms such as insurance schemes or agricultural subsidies. Large parts of tropical agriculture still operate in a range where such 'externalised' risk management options do not exist and where thus a choice for monocultures carries unaffordable risks. At the farm level ecosystem resilience can be extended beyond resources maintained on farm or in the accessible neighbourhood by being part of a larger agricultural production and germplasm delivery system.

Ewel (1986) and Moreno and Hart (1979) are among those who have advocated using plant functional groups as a basis for the (plot level) design of multi-plant agroecosystems. These designs also rely, explicitly or implicitly, on the impact that the effect of increasing the diversity of the vegetation system will have in enhancing the associated biodiversity both above- and below-ground and thence the probability of maintaining ecosystem services over a wider range of stress and disturbance. The evidence comparing such systems is almost entirely, however, based on assessments of yield, Vandermeer et al. (1998) reviewed the literature on inter-cropping of all types and concluded that yield gains in comparison with monocrops depends on the specific complementarities in resource use and seasonal development of the components. As risks for the farmer depend on farm level diversity of potentially productive resources rather than on plot-level diversity, the focus of much agroecological research may have been too narrow.

Another key aspect that needs to be changed is the continuing separation of different aspects of management interventions on the base of disciplinary experience, such as soil or nutrient management from pest management. Interventions to ameliorate the impacts on any one of the different ecosystem services (as well as on productivity) are likely to influence others. Practices targeted at productivity but well documented in terms of their supportive, ameliorative or regenerative effect on other ecosystem services should be a top priority.

9. Does the relationship between diversity and ecosystem services change across scales?

Almost all the evidence that exists for the relationship between diversity and function in agroecosystems concerns the plot (and often the micro-plot or laboratory chamber) scale. But in order to provide policy makers with appropriate advice on the functional value of diversity it is necessary to consider the ways in which the three factors we have been considering—biodiversity, agricultural productivity and profitability, and ecosystem services—intersect at the landscape scale. Whilst the inter-relationships that we have described at the plot (patch) scale may help in understanding what happens at the landscape scale there is also the possibility that the rules change across spatial scales. The productivity of any land-use system can be expressed on an area basis and the aggregate productivity across a landscape on the basis of the fractions occupied by different land uses. Biodiversity, however, has more complex scaling relationships and cannot simply be aggregated in this way. Nor can many of the functions that have been discussed here.

Much of the diversity in a landscape may exist at scales beyond the farm (between-farm variability being larger than within-farm diversity), and the dynamics of diversity thus depend on the degree to which different farms remain (or become more) different. As agricultural research and extension have been based on the economies of scale that are perceived as attainable by homogenisation of farms with similar demands for inputs and services and similar outputs for markets, the trend in agricultural intensification has often resulted in the reduction of inter-farm diversity. The green revolution provides a good illustration of this process which is generally supported by policy interventions that tend to promote homogeneity in farmer goals, practice and behaviour, at least over the short term. The agents of change in biodiversity beyond farm level are essentially different from those on farm.

In Fig. 2 we hypothesise that the relationship between species richness and specific ecosystem services at the landscape scale may follow a relationship analogous with that of the Vitousek–Hooper model—together of course with all the attendant qualifications. That is to say that ecosystem services at the landscape scale are optimised by a diversity of land uses, but the number that are required for optimisation is relatively small. If the hypothesis is correct then it would suggest that the presence of a relatively small number of different land-use types should be sufficient to satisfy the functional needs of the majority of ecosystem services. This generality needs, however, to be detailed for any given landscape into specifics with respect to not only the types but also their sizes, shapes, their patterns and location on the landscape and practices of management.

It can be further hypothesised that at the higher scales of landscape and region the frequency and intensity of disturbance and stress (both natural and anthropogenic) is greater than those at the plot or farm scale and increasingly beyond the control of the land users. Prevention of decreases in the stability of agroecosystems and management of restoration become more difficult and costly and eventually become impossible from both biological and economic perspectives because connectivity is too high and disturbances too large. The ecosystem services that enhance the resilience and adaptation of systems, such as biodiversity, thus become more and more important a feature of sustainable management as the scale of operation widens.

Fig. 3 hypothesises a number of relationships implied in the above discussion. We have argued that at the plot and farm scales individual land managers and farmers manage biodiversity largely through simplification (i.e., by decreasing connectivity and maintaining agroecosystems at a stage of early succession) and substitution. Decreases in connectivity may, under specific conditions reach a threshold level of irreversibility, in which case the agroecosystem loses its resilience. However, the individual land user can in most cases manage and control agroecosystem disturbances and stresses, such as pest outbreaks or sudden changes in relative prices, by making adjustments in the management of resources (land, water, germplasm, knowledge, labour, capital) at the farm scale.

Curves 1 and 2 of Fig. 3 deal with this case of diversity management at the plot to farm (i.e. land-use) scale and are therefore concerned with alpha diversity—that is, within these boundaries. (Curves 3 and 4 refer to higher landscape scales and are discussed in the next section.) The arrow linking Curve 1 to Curve 2 represents the capacity of farmers to maintain the ecosystem services necessary for their production goals whilst sacrificing diversity. This shift thus hypothesises that at the plot and farm scale management interventions can compensate for losses of diversity, although of course both the economic and of-site ecological consequences of this remain unstated and will be very circumstantial. We know, as shown for small-scale farms in Kenya by Osgood (1998), that many farmers do value genetic and species diversity on their farms, as they are aware that it minimises economic risk by enhancing on-farm diversification of plant and animal production. The history of agriculture provides many examples of how even extreme reductions in biodiversity can be managed, through periods of disturbance, by individual land users by substitution (e.g. chemicals, labour). Therefore, even though biodiversity has important ecological functions at the farm scale, it is nevertheless possible to decrease biodiversity levels very substantially at that scale while maintaining the productivity and resilience of agroecosystems. We hypothesise below, however, that at higher scales the control and management of disturbances and stresses becomes more and more problematic and costly and the resilience function of biodiversity thus becomes an increasingly important issue in management.

9.1. *Keep it simple: maintain ground cover*

We have already emphasised the over-arching influence of the plant cover and diversity on the associated functional diversity and thence on the properties of resilience. The simplest rule for managing landscapes is thus to say that if the vegetation is diverse then the associated diversity and functions will be taken care of. The immediate implication of this is that monotypic landscapes—vast areas of the same crop or livestock system—are likely to be the most vulnerable to the same dangers to ecosystem services pictured earlier for the farm or plot scale. Examples of these effects are the pollution of ground water by nitrates and pesticides in large-scale chemical-based agriculture and the difficulty of controlling epidemics in genetically homogeneous stands of vast area. These, however, seem simply to be the same issues as those at the plot scale only writ larger. The mechanism for correction generally proposed is that of diversification of the type of land-use system in space and time. What are the consequences that may flow from this?

9.2. Landscape mosaics

The majority of agricultural landscapes in the tropics, in contrast with most of the northern temperate zones, are mosaics of different land uses. How does this influence the biodiversity–ecosystem service relationship? At the plot scale the ecosystem services which is probably the most sensitive to biodiversity loss is the biological pest control system. The management opportunities for this increase with widening scale as greater opportunity for diversity in both genetic signals and physical structure of the vegetation permit a wider diversity and larger reservoir of control organisms. Similarly many of the endangered invertebrates and microorganisms of the soil community are mobile, or may be carried by vectors, and can thus recolonise degraded areas from within mosaics that provide suitable reservoirs. Others (e.g. earthworms) are less so, however, and re-inoculations may be necessary. In each of these cases the size, pattern of arrangement and rotation in time of land uses on the landscape will have significant effect on the efficiency of ecosystem service provision. Management at the landscape scale offers greater opportunity than at the plot and farm for varying land-use over time. Izac and Swift (1994) argued that sustainable land management could most easily be achieved at this scale by means of balance between aggrading and degrading areas, i.e. between patches of high exploitation and those of fallow or rest, in contrast to advocacy of high protection and diversity over the entire landscape. Soil organic matter change is a specific and far-reaching example. In areas of intensive production and harvest the soil carbon content may decrease but under fallow or tree-based production it can be re-built. The balance between these two options affect nutrient cycling, soil structure, water regimes and the emission of greenhouse gases. The policy requirements for such integrated management of landscape mosaics are, however, very different to the production-related approaches that currently prevail in favour of landscape homogenisation.

The third hypothesis of Vandermeer et al. (1998) predicts that a higher diversity of species will be required to provide a buffer against stress and disturbance at the landscape scale than will be the case for any single patch within it (i.e. gamma diversity will be higher than the sum of alpha diversity). This is pictured in Fig. 3 by the difference between Curves 1 and 3. Humans can intervene relatively easily (although not necessarily cost-effectively) at the plot scale to substitute for diversity loss—as represented by the difference between Curves 1 and 2. At the landscape scale, however, intervention by humans, including these substitutive actions, will tend to widen the range of stress and increase the frequency of disturbance. We therefore hypothesise that this will result in yet greater need for diversity to ensure the maintenance of ecosystem services and resilience. This is shown by the arrow linking Curves 3 and 4 in Fig. 3.

Substitutive management for purposes of restoring ecosystem services (i.e. to achieve a shift back from Curve 4 to Curve 3, analogous to the Curve 1 to 2 shift in Fig. 3) is likely to be both technically difficult and prohibitively expensive at this scale and may suffer from a ‘free rider’ problem where it is difficult to get all beneficiaries to share the costs. We contend therefore that the implication of this hypothesis is of the very high risk associated with ignoring landscape scale management and focussing only on policies that promote plot scale interventions. Plot scale activities are more likely to exacerbate landscape scale problems than repair them. On the other hand, landscape scale interventions offer great opportunity for improvements at the plot scale by increasing overall integration and resilience. There is thus more functional justification for arguing in favour of maintaining or enhancing biological diversity at the landscape scale than there is at the scale of the plot.

This model is of course simplistic and does not provide any guide to other features such as the size, shape and position (pattern) of patches on the landscape or on the temporal relationships between them. The hierarchical relationship between ecosystem services should assist in developing rules for these aspects. The regulation of erosion and water flows operates at a higher level in the hierarchy of controls than do aspects of nutrient cycling, soil structure and gas emissions or pest controls. The next part of this volume takes up these higher level aspects of landscape management under the title of 'watershed services'. The lower level services such as nutrient cycles and biological control activities may then be built in through focus on aspects such as the degree of connection between the patches and the location, direction and intensity of the flows between them. It may be useful to classify land-use types into 'functional groups' in a manner analogous with that for species in order to develop more meaningful relationships between diversity and function at the landscape scale.

10. Policy implications

The changes associated with agricultural intensification, including the attendant processes of diversity reduction and substitution of function, are made in response to food need, market opportunity, and perceptions of increased management efficiency associated with specialisation. These factors remain a dominant reality within market-oriented agriculture where a small number of specific products have high value and specialisation thus becomes a desirable target. Van Noordwijk and Ong (1999) discussed the paradox that urban consumers have access to an increasingly diverse array of food resources that are produced on specialised farms of greatly reduced internal diversity. Observed changes in diversity at one scale may thus not represent changes at other levels. The risks to agroecosystem services of simplifying ecosystems and substituting biodiversity by labour and chemicals (e.g., in pest control) are those of losing some keystone functions including the ability of an agroecosystem to adapt to change without yet further substitutive interventions. The evidence, as briefly described above, that ecosystem services might be significantly impaired in agroecosystems as intensification increases is substantial although the role of biodiversity is far from clearly understood. The farmer may not perceive these effects to be serious if the economic environment enables continuing profit based on subsidies related to the substitution process, within markets that do not price environmental services or externalities. This has been the basis of agricultural development in Europe and North America for many decades. It thus appears that in the absence of specific policy interventions, to attain profitability, even without petro-chemical substitution, agroecosystem diversity is likely to be kept low. Associated with this low diversity there is a risk of crossing threshold levels for the maintenance of ecosystem services the restoration of which is likely to be extremely costly, let alone feasible. Decisions about the management of agroecosystems in market economies do not normally take into consideration the costs of interfering with ecosystem services, including those in which biodiversity plays a strong influence. But when agroecosystems are driven across thresholds from a desired to an undesirable state, the costs to society of being in this new undesirable state, or of restoration of a more desirable one if it is feasible, can be extremely high. Therein lies the risk of simplifying ecosystems. Holling (1986) provided a seminal analysis of the consequences of a number of such irreversibilities.

Policies for sustainable agriculture, i.e. to promote integrative practices that focus on the conservation of resources (including genetic diversity) as well as productivity, have proved elusive. If the policy needs are extended to include the management of biodiversity at the landscape scale in order to protect and enhance a wide range of ecosystem services, the problem becomes more acute. There are two particular reasons why the problem is exacerbated at higher scales. First, population pressure and globalisation of trade and the concomitant land-use changes (expansion of cities into agricultural lands and of agriculture into marginal areas) result in increased frequency and intensity of disturbances and stresses by comparison with those at the farm scale. The capacity to correct these effects also diminishes because the sensitivity of the systems increases in concert with their connectivity as one moves up the hierarchy of scales (Holling, 1986).

Second, the higher the scale under consideration, the more difficult it is for the increased numbers of individual land users to develop an effective management strategy for agroecosystem disturbances that takes ecological interactions and connectivity into consideration. Even at the scale of small watersheds, it is not often the case that land users have been successful in developing collective and effective means of control and management of disturbances. Furthermore, even if these land users have full knowledge of the relevant level of connectivity necessary to ensure resilience at the watershed scale, different sectors of society place differing levels of importance on ecosystem services and diversity. Farmers in tropical countries are unlikely to place as high a value on these functions of landscape diversity as does the community at large or the national society. They are furthermore highly unlikely to value the serependic (i.e. future) value of diversity, which is much more likely to be valued by national and global communities.

In economic terms, farmers value some of the on-farm benefits of diversity and very few of the off-farm benefits, for the usual reasons that costs and benefits outside of the managers' domain (i.e. externalities) are generally not taken into account by individual decision-makers. The argument is, however, not simply about off-farm effects of biodiversity being ignored. Farmer knowledge varies greatly. There may be many on-farm ecosystem services of which farmers are unaware (e.g., the role of microorganisms), and thus cannot value, as well as services they may be aware of but will not consider important (e.g., reduction of greenhouse gas emissions). The same services may be valued by other groups in society, with a different perspective and set of interests. What is a beneficial service for one group may also be a cost for another (e.g. the perception of earthworms as 'pests' for paddy rice farmers, the trade-off between carbon sequestration and SOM mining). For these reasons, management of ecosystem services, and of biodiversity at the landscape scale, as well as management of disturbances in agroecosystems in land-use mosaics, is unlikely to be optimal, from either an ecological or an economic perspective, in the absence of specific policy or institutional interventions. Lack of knowledge of threshold levels in connectivity at different scales, different perspectives on the value of biodiversity, externalities and difficulties in large groups of land users coming together in developing effective means of controlling disturbances at the landscape scale thus result in biodiversity being managed by individual farmers in a sub-optimal manner.

We therefore conclude, on the basis of the relationships we have hypothesised earlier, that it will prove very costly to manage ecosystem services at the watershed, landscape and higher scales unless the functional value of biodiversity for productivity at the plot and farm scale and its interaction with 'externalities' beyond are perceived and valued. Furthermore, unless in particular the role of biodiversity in enhancing resilience is understood and factored into effective policy or institutional interventions, ecosystem diversity is unlikely to be maintained at the landscape scale without deliberate policy interventions at national and sub-national levels which take into account the real value of maintaining ecosystem services, given the externalities they generate and given their contribution to resilience. The biggest challenge is in the realisation that most of diversity as well as much of its positive role in resilience probably exists beyond the farm scale, and that thus diversity of management decisions by farmers rather than any specific management system is key to its maintenance in the landscape. Assessments of biodiversity values of different management scenarios will have to form the basis of discussions of the effectiveness of different policy interventions. These policy implications and the need for diversity enhancing communal action remain largely unexplored territory.

Finally, the absence of clear evidence should not be taken as evidence for the absence of effects and thus as a reason for doing nothing. Some economists have proposed that, in view of our relatively poor understanding of the exact roles of biodiversity in ecosystems on the one hand and of the potentially devastating effects of biodiversity loss on the other hand, a precautionary principle should be used in managing diversity. This principle acknowledges that while we may not be able to justify what some see as redundant species, there may be an extinction threshold that would result in an unacceptable level of ecosystem failure. Consequently, extreme care and precaution must be taken, and it is preferable to err on the conservative side (Perrings, 1991). The precautionary principle introduces an important concept, namely that of the risk of managing agroecosystems in such a way that threshold levels of biodiversity loss in relation to ecosystem services are ignored. The 'risk premium' that the precautionary principle suggests is hard to quantify as yet.

11. Concluding remarks

In the above discussion we have quoted or proposed a range of hypotheses concerning the relationships between biological diversity and ecosystem functions, and their implications for the management of agricultural landscapes. The general relationships that have been proposed may have to be replaced by more specific hypotheses of the relation between components of overall biodiversity and specific environmental functions, bounded in space and time. Sweeping generalisations from experiments that are necessarily restricted in space and time, and for example, do not include major parts of the diversity-generating processes (including 'lateral flows' of dispersal and migration for re-establishment), are unlikely to be helpful in guiding the development of agroecosystems that have to provide for short, medium and long-term service functions. Future investigations should utilise co-evolved communities, be structured to investigate the distinct roles of clearly defined functional groups, separate the effects of between- and within-group diversity and be conducted over a range of stress and disturbance to identify threshold levels of irreversibility of functional losses. This might include: testing the basic functional-biodiversity rule by experimentally determining the minimal level of diversity between and within functional groups that is necessary to maintain productivity, integrity and perpetuation of ecosystems; characterising the functional groups of organisms necessary to maintain specific ecosystem services; determining the ecosystem function and service effects that ensue from elimination or substitution of key functional groups, including particular investigation of controls over below-ground diversity and function exerted by particular plant functional groups and other keystone organisms; and determining (and developing indicators for) the biodiversity thresholds for different ecosystem services. An interesting extension of the latter study might be to investigate whether similar thresholds exist for the intrinsic, utilitarian and serependic values of biodiversity.

Society as a whole has an interest in ecosystem services that are manifested substantially at scales above that of the field, plot or farm. At the scale of the watershed or landscapes there is, in comparison with any single patch, a greater range of environmental stress and higher frequency of disturbance, including of extreme events. The maintenance of ecosystem services at these scales thus requires either a higher diversity of species within functional groups or a greater investment in substitutive management to maintain ecosystem services. These increments in diversity and/or investment are unlikely to be simply additive in view of the significant shifts in complexity that occur with shifts across scale. Optimal maintenance of ecosystem services at the landscape scale may be most readily achieved by a mosaic of a relatively few land-use types. This model is, however, likely to be overly simple because of: (a) differences in functional impact of different land-use types and (b) the importance of organisation at the landscape scale in terms of the size, shape and location pattern of the constituent land uses.

In developing appropriate land-use scenarios landscapes should be compared with respect to the aggregate values of their component land uses for intrinsic, utilitarian and functional (ecosystem service) values of biodiversity. This would be assisted by establishing a typology of land uses in terms of their efficiency in maintaining ecosystem service and in the trade-offs between this and profitability. The results of the ASB project provide a model for this approach with respect to the interactions between carbon sequestration potential and profitability. The relative costs and benefits of segregating the intrinsic, utilitarian and functional uses of biodiversity between different land-use or landscape units compared with integrating them within such units is another parameter that should be of significant value for policy development.

This review confirms two unsurprising but crucial elements for policy development: first, that whilst a number of important analogies can be drawn across scales with respect to the management of the relationships between biodiversity and ecosystem services, there are also emergent properties that necessitate different approaches; second that the value placed on the relationship between biodiversity and function (ecosystem services) by individual land users is markedly different than those perceived by the community at different levels of society. We have indicated a number of biological and socio-economic issues that need to be clarified in order to provide more explicit advice to policy makers. No single optimal value can be placed on the biodiversity within a landscape. Land-use decisions are likely to be optimised if decision-makers can be provided with scenarios showing how various land-use combinations result in different levels of diversity and the efficiency of different ecosystem services, and the associated values of biodiversity. In so-doing it will be important to include aspects of temporal change as well as pattern on the landscape as both these factors influence the resilience of the landscapes which should be regarded as a factor of over-riding importance. These scenarios can then be used to identify policy interventions and institutional arrangements necessary to achieve the desired objective, whether it is one dominated by agricultural productivity targets or the maintenance of ecosystem services or the conservation of biodiversity, or a combination of all three.

Appendix A. Key functional groups: a preliminary classification

We have defined a functional group in the text as 'a set of species that have similar effects on a specific ecosystem-level biogeochemical process'. There are many examples of classification of species in this way within specific taxonomic or trophic groups (e.g. for plants or pests). There is no single classification to suit all purposes. In each case it is clear that the number of functional groups that is recognised, the criteria that are used to classify them and the degree of subdivision that is applied is a function of the question that is being addressed. We propose here a classification into the 10 major groups that are briefly described below, together with such subdivision as may be necessary, for the purposes addressed in this paper, i.e. the relationships between biodiversity and function with particular respect to agriculture and ecosystem services. These Key Functional Groups are listed in Table 1 in relation to the ecosystem services they provide. The relationships between them are shown in Fig. 4. We suggest that this could provide a useful framework for investigating and testing key questions on this topic. A hierarchical structure is suggested (Fig. 4). At the highest level are four major categories related to major trophic functions at the ecosystem scale, i.e. Primary Production, Primary Regulation, Service Provision and Secondary Regulation. At the next level are the 10 groups listed in Table 1 that perform distinct ecosystem functions; and at the third level are subdivisions which it may be functionally and/or taxonomically useful to distinguish (e.g. vertebrate grazers versus invertebrate pests among the herbivores).

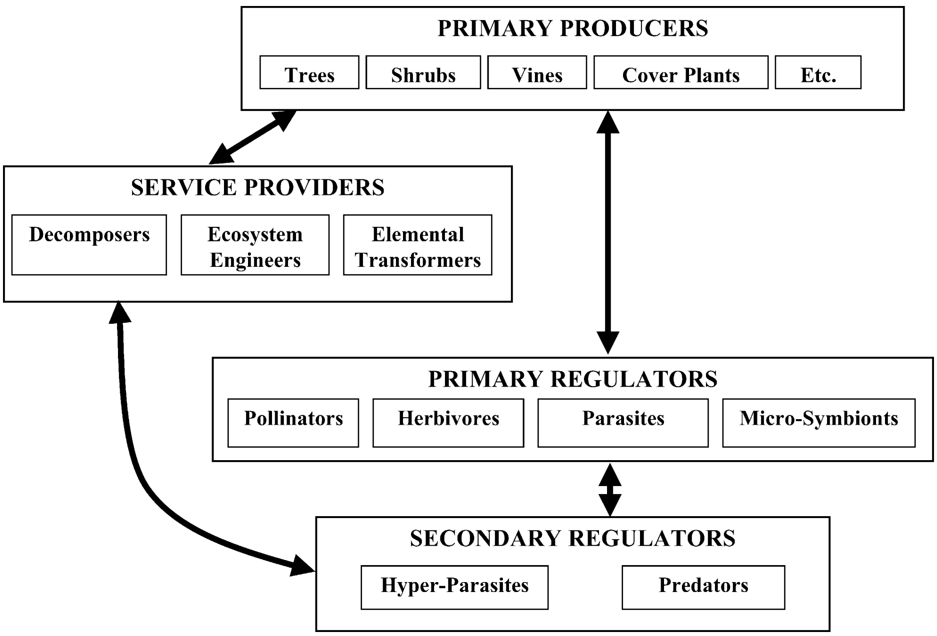

Fig. 4. Hierarchical relationships between different categories of functional group, see Table 1 and related footnotes.

Table 1
Relationship between key functional groups of organisms, the ecosystem level functions they perform and the ecosystem goods and services they provide

Ecosystem goods and services	Ecosystem functions	Key functional groups
Ecosystem goods including		
Food	Primary and secondary (herbivore) production	Plants, vertebrate herbivores
Fibre and latex	Primary production and secondary metabolism	Plants
Pharmaceuticals and agro-chemicals	Secondary metabolism	Plants, bacteria and fungi (decomposers, etc.)
Ecosystem services including		
Nutrient cycling	Decomposition	Decomposers
	Mineralisation and other elemental transformations	Elemental transformers
Regulation of water flow and storage	Soil organic matter synthesis	Decomposers
	Soil structure regulation—aggregate and pore formation	Ecosystem engineers
Regulation of soil and sediment movement	Soil protection	Plants
	Soil organic matter synthesis	Decomposers
	Soil structure maintenance	Ecosystem engineers
Regulation of biological populations including diseases and pests	Plant secondary metabolism	Plants
	Pollination	Pollinators[a]
	Herbivory	Herbivores[a]
	Parasitism	Parasites[a]
	Micro-symbiosis	Micro-symbionts[a]
	Predation	Hyper-parasites[b], predators[b]
De-toxification of chemical or biological hazards including water purification	Decomposition	Decomposers
	Elemental transformation	Elemental transformers
Regulation of atmospheric composition and climate	Greenhouse gas emission	Decomposers, elemental transformers, plants, herbivores

Primary production: In some ecosystems photosynthetic microorganisms may constitute as significant group, e.g. rice ecosystems). Here we deal only with plants. *Plants*. There is a long history of classification of plants into functional groups. The groupings have been based on a variety of reproductive, architectural and physiological criteria. For the purposes of this paper the efficiency of resource capture is suggested as the main criterion. This will be determined by features of both architecture (e.g. position and shape of the canopy and depth and pattern of the rooting system) and physiological efficiency. A very simple classification could for instance distinguish the roles of trees, shrubs, vines and cover plants, etc. and then subdivisions within each of these groups. Much more detailed consideration of these aspects is given by Smith et al. (1997).
Decomposers: This is a group of great diversity which can be subdivided taxonomically (bacteria, fungi, invertebrates, etc.) and in relation to size both of which correlate somewhat with functional roles in the breakdown (e.g. detritivorous invertebrates) and mineralisation (fungi and bacteria) of organic materials of plant or animal origin (Swift et al., 1979; Lavelle and Spain, 2001).
Ecosystem engineers: These are organisms that change the structure of soil by burrowing, transport of soil particles and formation of aggregate structures. The term is often confined to the macrofauna such as earthworms and termites but fungi and bacteria also play a key role in the binding of soil aggregates. Many of these organisms also contribute to the processes of decomposition.
Elemental transformers: This may be the most diverse group of all and deserving of substantial subdivision. It includes a range of autotrophic bacteria that utilise sources of energy other than organic matter (and therefore not classifiable as decomposers) that play key roles in nutrient cycles as transformers of C, N, S, etc. In addition there are heterotrophs that thus have a decomposer function but also carry out elemental transformations beyond mineralisation (e.g. free-living di-nitrogen fixers).

[a] Primary regulators—These are a set of functional groups which have a significant regulatory effect on primary production and therefore influence the goods and services provided by the plants. *Pollinators*. This is a taxonomically very disparate group of organisms including many insect groups and vertebrates such as birds and bats. However, there does not appear to be any generally accepted categorisation based on feeding behaviour or similar criteria (Barbara Gemmill, pers. commun.). *Herbivores*. A great variety of organisms feed directly on primary producers. Vertebrate grazers and browsers are readily distinguished from invertebrate pests although their impacts on the plants may have similar functional significance at the ecosystem level. Each of these major groups are subdivisible in terms of, for instance, feeding habits. The balance between different types of browser, for instance, can influence the structure of the canopy. *Parasites*. Microbial infections of plants may limit primary production in analogous manner to herbivory. Parasitic associations can also influence the growth pattern of the plants and thence their architecture and physiological efficiency. *Micro-symbionts*. There is a wide range of microbial infections that are beneficial rather than destructive of which the most familiar are di-nitrogen fixing bacteria and mycorrhizal fungi. Service provision—The functional groups within this category also strongly influence primary production but not in the directly destructive or stimulatory way of the primary regulators. They also provide a set of ecosystem services distinct to those deriving mainly from the primary producers.

[b] Secondary regulators—*Hyper-parasites and predators*. This is diverse group of microbial parasites and vertebrate and invertebrate predators that feed on decomposers, herbivore, pollinators, etc. They have particular significance in agriculture because of the service of biological control of pests and diseases that they play.

Further levels of subdivision may also be useful or necessary in some cases.

References

Allen, T.F.H., Starr, T.B., 1982. Hierarchy: Perspectives for Ecological Complexity. The University of Chicago Press, Chicago.

Barrett, S., 1993. Optimal economic growth and the conservation of biological diversity. In: Barbier, E.B. (Ed.), Economics and Ecology: New Frontiers and Sustainable Development. Chapman & Hall, London, pp. 130–145.

Beare, M.H., Coleman, D.C., Crossley Jr., D.A., Hendrix, P.F., Odum, E.P., 1994. A hierarchical approach to evaluating the significance of soil biodiversity to biogeochemical cycling. Plant Soil 170, 5–22.

Beare, M.H., Reddy, M.V., Tian, G., Srivastava, S.C., 1997. Agricultural intensification, soil biodiversity and agroecosystem function in the tropics: the role of decomposer biota. Appl. Soil Ecol. 6, 87–108.

Chauvel, A., Grimaldi, M., Barros, E., Blanchart, E., Sarrazin, M., Lavelle, P., 1999. Pasture degradation by an Amazonian earthworm. Nature 389, 32–33.

Conway, G.R., 1993. Sustainable agriculture: the trade-offs with productivity, stability and equitability. In: Barbier, E.B. (Ed.), Economics and Ecology: New Frontiers and Sustainable Development, pp. 46–65.

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

Ewel, J.J., 1986. Designing agricultural ecosystems for the humid tropics. Ann. Rev. Ecol. Syst. 17, 245–271.

Ewel, J.J., 1999. Natural systems as models for the design of sustainable systems of land use. Agrofor. Syst. 45, 1–21.

Ewel, J.J., Mazzarino, M.J., Berish, C.W., 1991. Tropical soil fertility changes under monocultures and successional communities of different structure. Ecol. Appl. 1, 289–302.

GCTE, 1997. Effects of global change on multi-species agroecosystems. In: Swift, M.J., Ingram, J.S.I. (Eds.), Implementation Plan, Report No. 13. GCTE, Wallingford, UK.

Giller, K.E., Beare, M.H., Lavelle, P., Izac, A.-M.N., Swift, M.J., 1997. Agricultural intensification, soil biodiversity and agroecosystem function. Appl. Soil Ecol. 6, 3–16.

Griffin, D.M., 1972. Ecology of Soil Fungi. Syracuse University Press/Chapman and Hall, Syracuse, NY, USA.

Grime, J.P., 1997. Biodiversity and ecosystem function: the debate deepens. Science 277, 1260–1261.

Heywood, V.H., Bates, I., 1995. Introduction. In: Heywood, V.H., Watson, R.T. (Eds.), Global Biodiversity Assessment. Cambridge University Press, UNEP, pp. 1–21.

Hodgson, J.G., Thompson, K., et al., 1998. Does biodiversity determine ecosystem function? The Ecotron experiment reconsidered. Funct. Ecol. 12, 843–848.

Holling, C.S., 1973. Resilience and stability of ecological systems. Ann. Rev. Ecol. Syst. 4, 1–23.

Holling, C.S., 1986. The resilience of terrestrial ecosystems: local surprise and global change. In: Clark, W.C., Munn, R.E. (Eds.), Sustainable Development of the Biosphere. Cambridge University Press, Cambridge, pp. 325–347.

Hooper, D., Vitousek, P.M., 1997. The effects of plant composition and diversity on ecosystem processes. Science 277, 1302–1305.

Hubbell, S.P., 2001. The Unified Neutral Theory of Biodiversity and Biogeography. Monographs in Population Biology 32. Princeton University Press, Princeton (USA) and Oxford (UK).

Izac, A.M., Swift, M.J., 1994. On agricultural sustainability and its measurement in small scale farming systems in sub-Saharan Africa. Ecol. Econ. 11, 105–125.

Joshi, R.C., Matchoc, O.R.O., Cabigat, J.C., James, S.W., 1999. Survey of earthworms in the Ifugao rice terraces, Philippines. J. Environ. Sci. Manage. 2, 1–12.

Kahindi, J.H.P., Woomer, P., George, T., Moreira, F.M., Karanja, N.K., Giller, K.E., 1997. Agricultural intensification, soil biodiversity and agroecosystem function in the tropics: the role of nitrogen-fixing bacteria. Appl. Soil Ecol. 6, 87–108.

Kerry-Turner, R., 1999. Environmental and ecological economics perspectives. In: van den Bergh, J.C.J.M. (Eds.), Handbook of Environmental and Resource Economics. Edward Elgar, Cheltenham, UK, pp. 1001–1033.

Kinzig, A.P., Pacala, S., 2001. Successional biodiversity and ecosystem functioning. In: Kinzig, A., Pacala, S.W., Tilman, D. (Eds.), The Functional Consequences of Biodiversity, Empirical Progress and Theoretical Extensions. Princeton University Press, Princeton and Oxford, pp. 175–212.

Lavelle, P., Spain, A.V., 2001. Soil Ecology. Kluwer Academic Publishers, London.

Lawton, J.H., Brown, V.K., 1993. Redundancy in ecosystems. In: Schultze, E.D., Mooney, H.A. (Eds.), Biodiversity and Ecosystem Function, Ecological Studies, vol. 99. Springer-Verlag, Berlin, pp. 255–268.

Lawton, J.H., Naeem, S., et al., 1998. Biodiversity and ecosystem function: getting the Ecotron experiment in its correct context. Funct. Ecol. 12, 848–852.

Matson, P.A., Parton, W.J., Power, A.G., Swift, M.J., 1997. Agricultural intensification and ecosystem properties. Science 277, 504–509.

Mooney, H.A., Lubchenco, J., Dirzo, R., Sala, O. (Eds.), 1995. Biodiversity and Ecosystem Function: Ecosystem Analyses. In: UNEP Global Biodiversity Assessment, Section 6. Cambridge University Press, Cambridge.

Mooney, H.A., Cushman, J.H., Medina, E., Sala, O.E., Schulze, E.D. (Eds.), 1996. Functional Roles of Biodiversity: A Global Perspective. In: Proceedings of the SCOPE 55. Wiley, Chichester.

Moreno, R.A., Hart, R.D., 1979. Intercropping with cassava in Central America. In: Weber, E., Nestel, B., Campbell, M. (Eds.), Intercropping with Cassava. IDRC, Ottawa.

Naeem, S., 2000. Reply to Wardle et al. Bull. Ecol. Soc. Am. 81, 241–246.

Naeem, S., Li, S., 1997. Biodiversity enhances ecosystem reliability. Nature 390, 507–508.

Naeem, S., Thompson, L.J., Lawler, S.P., Lawton, J.H., Woodfin, R.M., 1994. Declining biodiversity can alter the performance of ecosystems. Nature 368, 734–737.

Neutel, A.M., 2001. Stability of complex food webs: pyramids of biomass, interaction strengths and the weight of trophic loops. Ph.D. Thesis. University of Utrecht, The Netherlands, 128 pp.

Nunes, P.A.D., van der Bergh, J.C.J.M., 2001. Economic valuation of biodiversity: sense or nonsense? Ecol. Econ. 39, 203–222.

Osgood, D., 1998. Valuing Biodiversity: Evidence from Farming Households in Mwea Kenya. Ph.D. Thesis. University of London.

Perrings, C., 1991. Reserved rationality and the precautionary principle: technological change, time and uncertainty in environmental decision-making. In: Costanza, R. (Ed.), Ecological Economics: The Science and Management of Sustainability. Columbia Press University, New York.

Pottinger, A.J., Burley, J., 1992. A review of forest biodiversity research in Africa. J. Trop. For. Sci. 5, 291–307.

Schultze, E.D., Chapin, F.S., 1987. Plant specialization to environments of different resources availability. Ecol. Stud. 61, 120–148.

Schulze, E.A., Mooney, H.A. (Eds.), 1993. Biodiversity and Ecosystem Function. Springer-Verlag, Berlin.

Smith, T.M., Shugart, H.H., Woodward, F.I., 1997. Plant Functional Types: Their Relevance to Ecosystem Properties and Global Change. Cambridge University Press, Cambridge.

Swift, M.J., 1976. Species diversity and the structure of microbial communities in terrestrial habitats. In: Anderson, J.M., MacFadyen, A. (Eds.), The Role of Aquatic and Terrestrial Organisms in Decomposition Processes. 17th Symposium British Ecological Society, Blackwells, Oxford, pp. 185–222.

Swift, M.J., 1999. Integrating soils, systems and society. Nat. Res. 35, 12–19.

Swift, M.J., Heal, O.W., Anderson, J.M., 1979. Decomposition in Terrestrial Ecosystems. Studies in Ecology 5. Blackwell Scientific Publications, Oxford, 372 pp.

Swift, M.J., Seward, P.D., Frost, P.G.H., Qureshi, J.N., Muchena, F.N., 1994. Long-term experiments in Africa: developing a database for sustainable land use under global change. In: Leigh, R.A., Johnson, A.E. (Eds.), Long-term Experiments in Agricultural and Ecological Sciences, pp. 229–251.

Swift, M.J., Vandermeer, J., Ramakrishnan, P.S., Anderson, J.M., Ong, C.K., Hawkins, B.A., 1996. Biodiversity and agroecosystem function. In: Cushman, J.H., Mooney, H.A., Medina, E., Sala, O.E., Schulze, E.D. (Eds.), Functional Roles of Biodiversity: A Global Perspective. Wiley, Chichester, pp. 261–298.

Swift, M.J., Anderson, J.M., 1993. Biodiversity and ecosystem function in agricultural systems. In: Schultze, E.D., Mooney, H. (Eds.), Biodiversity and Ecosystem Function. Springer-Verlag, Berlin, pp. 15–42.

Tilman, D., Downing, J.A., 1994. Biodiversity and stability in grasslands. Nature 367, 363–365.

Tilman, D., Wedin, D., Knops, J., 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. Nature 379, 718–720.

Tilman, D., Knops, J., Wedin, D., Reich, P., Ritchie, M., Sieman, E., 1997. The influence of functional diversity and composition on ecosystem processes. Science 277, 1300–1302.

Tilman, D., Lehman, C., 2001. Biodiversity, composition, and ecosystem processes: theory and concepts. In: Kinzig, A., Pacala, S.W., Tilman, D. (Eds.), The Functional Consequences of Biodiversity, Empirical Progress and Theoretical Extensions. Princeton University Press, Princeton, Oxford, pp. 9–41.

UNEP, 1995. In: Heywood, V.H., Watson, R.T. (Eds.), Global Biodiversity Assessment. Cambridge University Press, Cambridge.

Vandermeer, J., Van Noordwijk, M., Anderson, J., Ong, C., Perfecto, I., 1998. Global change and multi-species agroecosystems: concepts and issues. Agric. Ecosyst. Environ. 67, 1–22.

Van Noordwijk, M., Ong, C.K., 1999. Can the ecosystem mimic hypotheses be applied to farms in African savannahs? Agrofor. Syst. 45, 131–158.

Vitousek, P.M., Hooper, D.U., 1993. Biological diversity and terrestrial ecosystem biogeochemistry. In: Schulze, E.D., Mooney, H.A. (Eds.), Biodiversity and Ecosystem Function. Springer-Verlag, Berlin, pp. 3–14.

Wardle, D.A., Huston, M.A., et al., 2000. Biodiversity and ecosystem function: an issue in ecology. Bull. Ecol. Soc. Am. 81, 235–239.

Wardle, D.A., Nicholson, K.S., Booner, K.I., Yeates, G.W., 1999a. Effects of agricultural intensification on soil-associated arthropod population dynamics, community structure, diversity and temporal variability over a seven-year period. Soil Biol. Biochem. 31, 1691–1706.

Wardle, D.A., Yeates, G.W., Nicholson, K.S., Booner, K.I., Watson, R.T., 1999b. Response of soil microbial biomass dynamics, activity and plant litter decomposition to agricultural intensification over a seven-year period. Soil Biol. Biochem. 31, 1707–1720.

Woodward, F.I., 1993. How many species are required for a functional ecosystem? In: Schultze, E.D., Mooney, H.A. (Eds.), Biodiversity and Ecosystem Function, Ecological Studies, vol. 99, pp. 271–289.

Woodwell, G.M., Smith, H.H., 1969. Diversity and Stability in Ecological Systems. Brookhaven National Laboratory, Associated Universities, Inc., New York.

Yeates, G.W., Wardle, D.A., Watson, R.T., 1999. Responses of soil nematode populations, community structure, diversity and temporal variability to agricultural intensification over a seven-year period. Soil Biol. Biochem. 31, 1721–1733.

Available online at www.sciencedirect.com
SCIENCE DIRECT
Agriculture, Ecosystems and Environment 104 (2004) 229–244
Agriculture Ecosystems & Environment
www.elsevier.com/locate/agee

Environmental services and land use change in Southeast Asia: from recognition to regulation or reward?

Thomas P. Tomich [a,*], David E. Thomas [b], Meine van Noordwijk [c]

[a] ICRAF, PO Box 30677, Nairobi, Kenya
[b] ICRAF Chiang Mai, PO Box 267, CMU Post Office, Chiang Mai 50202, Thailand
[c] ICRAF SE Asia, PO Box 161, Bogor 16001, Indonesia

Abstract

Awareness of environmental services and land use change in Southeast Asia is high among scientists, policymakers, and society. In the case of transboundary smoke, the level of awareness and concern in the region is high, but subsides in between periods of 'crisis'. Although there is a rising level of awareness of habitat loss and associated loss of genetic diversity, the basic cause–effect relationships underlying the ecological roles of biodiversity are still debated. Degradation of watershed functions is the most mature of our three meso-scale environmental topics; indeed it shows signs of being 'fossilized' by vested interests in the present consensus. Land use planning and other regulatory approaches have had little success. Policy instruments for achieving meso-level environmental policy objectives through changing incentives such as payment schemes for environmental services, have not been tested widely in Southeast Asia (or anywhere else). Further research and experimentation needs to incorporate strategic consideration of processes and spatial scales of environmental impacts and resource governance.

Keywords: Land use change; Environmental services; Smoke; Biodiversity functions; Watershed functions; Environmental issue cycle; Southeast Asia

1. Environmental services: natural capital or human effort?

Once they were taken for granted or, if perceived at all, were viewed as free 'gifts of nature'. But by the beginning of the 21st Century, various forces—global and local; social, political and economic; climatic and ecological—have produced heightened awareness of degradation of environmental services in contemporary Southeast Asia. Landslides, flooding and smoke now figure regularly and prominently in the news media. Global climate change and loss of biodiversity remain rather abstract concepts, but loss of natural habitat and its consequence for 'flagship' species is evident to many. So, in the environmental 'issue cycle' (Tomich et al., this volume), the three specific topics of this collection (smoke, biodiversity, and watershed functions) would seem to be firmly established on the policy agenda and also in public awareness, poised for concrete actions toward mitigation of the associated problems.

The term 'environmental services' often is used as a generic concept. Yet, for any effective relationship between outside beneficiaries of these 'services' and the upland land use systems and communities that generate the services, it is necessary to be explicit in defining what the functions are, and how they can be measured

* Corresponding author.
Tel.: +254-20-524139/+1-650-833-6645;
fax: +254-20-524001/+1-650-833-6646
E-mail address: t.tomich@cgiar.org (T.P. Tomich).

0167-8809/$ – see front matter © 2004 Elsevier B.V. All rights reserved.
doi:10.1016/j.agee.2004.01.017

and monitored. So, we need to decompose the broad concept of 'environmental services' into constituent components in order to be clear on the cause–effect chains underlying the provision of services. To create appropriate incentives in order to effectively maintain or enhance these services, rewards must be properly directed toward those providing services. Conversely, to establish and maintain a willingness to pay on the side of the beneficiaries, a broadly shared and accurate understanding of the cause–effect relations is required.

Tomich et al. (this volume) introduced this collection by formulating a series of questions to help unpack perceptions and clarify whether environmental issues actually correspond to real 'causes' and important 'effects'—in other words, whether they constitute significant 'environmental services'. In all cases, measures to sustain or enhance environmental services require appropriate quantitative methods and data analysis. But, as encapsulated in the environmental 'issue cycle', the emphasis on evidence will shift from understanding causal pathways ('processes'), recognizing spatial extent and distribution ('patterns'), developing 'proxies' or 'indicators' for easy recognition and monitoring, and simplified yet accurate and validated measures that will facilitate negotiations among various groups, often with conflicting interests.

In the case of transboundary smoke, the level of awareness and concern in the region is high, but the private economic benefits of converting swamp forest to other land uses through slash and burn clearing methods and from the logging that often initiates a forest degradation cycle (which enhances fire susceptibility) are too great as yet to be effectively controlled in many countries. For biodiversity, the global existence values and option values of maintaining genetic diversity may be clear, but basic cause–effect relationships underlying the ecological roles of biodiversity are still debated. So, from a policymaker's perspective, it may not yet be clear what we should care about. Degradation of watershed functions is the most mature of our three environmental topics. It may even be 'over mature', as it shows signs of being 'fossilized' by vested interest in the present consensus, while challenges to the popularly held 'cause–effect' model of forests and watershed functions are resisted.

As we discuss in the concluding sections, regulation has been the conventional approach to mitigation of environmental problems, but some of the same forces that have produced heightened environmental awareness also have spawned an interest in positive alternatives in the form of incentive schemes to reward positive actions that maintain or increase the provision of environmental services. Pilot schemes, still in their early stages of testing, to reward upland farmers and communities for specific types of land use and land use change to sustain or enhance environmental services are a case in point. But in order to be effective and sustainable, these mechanisms need to fit within the changing governance landscape of Southeast Asia, and to be based on stronger and more spatially explicit biophysical research that can effectively assess and predict human impacts on environmental services at multiple spatial and temporal scales.

2. Managing smoke

As Byron (this volume) and others have shown, forest, land, and coal seam fires associated with drought and human activity are not new in Southeast Asia. But smoke problems are perceived to be worse than ever before, and that may well be true. From their historical review, Brookfield et al. (1995) concluded that '. . . the impact of drought and fire over the past 10 years has been much more devastating than at any time in at least the previous 100 years, and probably much longer'. And that assessment preceded the 1997/1998 crisis, which drew sustained international attention. (For comparative perceptions of the events of 1997/1998 by leading scientists working in Indonesian Borneo and the Brazilian Amazon, see interviews in Wuethrich (2000).)

Follow up investigations of the underlying causes of the 1997/1998 fires on Sumatra and Borneo has confirmed and extended the preliminary diagnoses put forward by Tomich et al. (1998) and Stolle and Tomich (1999). Remote sensing and social science investigations using participatory mapping conducted by a team of researchers from the Center for International Forestry Research (CIFOR), the World Agroforestry Centre (ICRAF) and the United States Forest Service (USFS), showed that both smallholders and large-scale plantations used fire as a tool, primarily for land clearing but also in specific contexts in extractive activities (Applegate et al., 2001; Dennis et al., 2004; Suyanto et al., 2002). For the first time, these social science investigations also have documented specific patterns in the use of fire as a weapon; arson arising from land disputes was shown to be an important albeit secondary factor. Finally, these detailed studies also confirm Vayda's (1998) observation that the incidence of accidental fires (fires that are set by smallholders for a purpose but which spread accidentally) may have been higher than is conventionally believed, particularly in Borneo. The importance of these accidents and the social context in which they occur in Northern Thailand is addressed in Hoare (this volume). Moreover, it has become clear that fires in the swamp forest zone produce a disproportionately large amount of smoke/haze per hectare burnt, and this ecological zone should thus receive specific attention (Murdiyarso et al., 2002).

It is encouraging that the CIFOR–ICRAF–USFS research on underlying causes has begun to discern meaningful patterns in the shape and extent of remotely sensed burn scars and to relate those burn scars to the underlying causes and broader environmental and social context documented in the social science studies (Dennis et al., 2004). This holds promise as a cheap and replicable tool for future fire forensic investigations, but it is not yet possible to attribute shares of the smoke problem between smallholders and large operators or arising from purposive burning (for land clearing, resource extraction, or arson) and accidents on the scale of large areas, such as Sumatra and Borneo.

Hoare's analysis (this volume) of the relative success of a provincial level fire and smoke management project demonstrates that it is not necessary to have a comprehensive national or regional analysis as a prerequisite to effective action. Hoare's documentation of significant incidence of accidental fires (and similar results for Indonesia) indicate a role for training programs in fire awareness and management. But the evidence from the CIFOR–ICRAF–USFS studies in Sumatra and Borneo, indicating that most fires are set deliberately, means that training alone will not be sufficient and that more fundamental attention to underlying driving forces, particularly the effects of insecure land tenure and property disputes (Dennis et al., 2004; Suyanto et al., 2002) will be necessary to manage the smoke problem. The private economic gains related to the causes of fire so far have not been effectively off-set by negative incentives that reflect the public costs.

Table 1
Estimates of fire and haze related damage (millions of US dollars)

Type of loss	Lost to Indonesia	Lost to other countries	Total
Timber	493.7	–	493.7
Agriculture	470.4	–	470.0
Direct forest	705.0	–	705.0
Indirect forest benefits	1077.1	–	1077.1
Capturable biodiversity	30.0	–	30.0
Fire fighting costs	11.7	13.4	25.1
Carbon release	–	272.1	272.1
Short-term health	924.0	16.8	940.8
Tourism	70.4	135.8	256.2
Other	17.6	181.5	199.1
Total fire and haze	3799.9	669.6	4469.5

Source: EEPSEA (1993–1998, p. 13).

The widely cited study by EEPSEA and WWF (1998) (Economy and Environment Program for Southeast Asia and World Wide Fund for Nature), estimated over $ 4.4 billion in damage from Indonesian fires and smoke in 1997 (Glover and Jessup, 1998, 1999). The EEPSEA/WWF estimates have been refined since by others, but that study was among the first and arguably was most influential in forming regional and global awareness of the smoke problem. A *global* environmental disservice, estimated as imputed value of carbon release of this event, was by far the biggest cost external to Indonesia considered in that study. After this global cost, those authors estimated that almost $ 3.8 billion (85%) was borne by Indonesia itself (Table 1). Although the situation in Singapore and Kuala Lumpur received most of the media attention, the EEPSEA/WWF estimates indicate that Indonesian citizens suffered the most short-term health effects by far ($ 924 million out of a total estimate of just over $ 940 million; about 98%). If this is the case, is this really primarily a regional problem? Balancing private gains of causing smoke and the public costs of the impacts should not require transboundary mechanisms, as the within-country net benefits would be sufficient, if effective institutional mechanisms could be found.

The impact of the EEPSEA/WWF study highlights the great time value of information. Above all else, efforts to publicize who is burning probably have the highest impact during a smoke emergency. Fortunately, remote sensing and the worldwide web provide powerful tools for doing that (see interview with Nabil Makarim in Tomich et al. (1999), also Tomich and Lewis, 2002).

Would further research to refine the influential EEPSEA/WWF cost estimates make a big difference to policymakers' perceptions or efforts to mitigate the problem? In other words, if more and better regional data on alternatives, causes, impacts and costs were available, would they likely prompt more effective national or regional action? The report on just such a science-policy activity by Murdiyarso et al. (this volume) does not provide much basis for optimism about effective *regional* action any time soon.

Fortunately, a grand regional strategy for Southeast Asia is not the only option to manage smoke—nor does this approach even seem to be the most obvious one given the insight that a big share of human costs are concentrated within the areas where the smoke problem originates. Hoare's study (this volume) of community and provincial level efforts in Northern Thailand moves longstanding interest in local institutional measures for management of fire and smoke from the anecdotal to the practical assessment of actual experience based on smoke mitigation interventions at this level. Even more fundamental is evidence from Indonesia (Applegate et al., 2001; Suyanto et al., 2002; Dennis et al., 2004) showing direct links between smoke and land tenure problems, thereby also establishing much more clearly how resolution of conflicts over property rights and resource access underpins a comprehensive approach to the problem.

Rhetoric, research and policy initiatives all would seem to have over emphasized comprehensive regional solutions to the detriment of efforts to identify, develop and replicate local approaches to the smoke problem. As Byron (this volume) stresses in his synthesis, a comprehensive solution rests fundamentally with local political accountability and local incentives to better manage fire and smoke. In other words, a comprehensive solution will be feasible when local government capacity and legitimacy exists concerning sanctions on those who reap the benefits of burning and when local government is accountable to those bearing the bulk of the costs. In the meantime, instead of ASEAN wide pronouncements, Hoare's study suggests that more could be accomplished (albeit incrementally) through efforts that begin by understanding local conditions, interests, and institutions. Not all fires are equal, and as Byron emphasizes, it makes sense to target big, smoky fires especially when climatic conditions already have adverse effects on air quality (also see Tomich and Lewis, 2002). Successful regulation of timing of burning to reduce smoke pollution in the Brazilian Amazon indicates that regulatory means of smoke management can minimize costs to land users while improving public health outcomes (Reinhardt et al., 2001), but that result also depends on even enforcement among rich and poor farmers, which in turn derives from some measure of broad-based local political accountability.

3. Distinguishing the two faces of biodiversity: existence and resilience

The concept of biodiversity has at least two distinct aspects or 'faces'. The global 'face' is charismatic, an international superstar, exemplified by the 'flagship taxa' of animals and plants that stimulate campaigns to prevent their extinction. Conservation of habitat to preserve this global biological heritage for future generations (of people) is a legitimate goal that inspires considerable public support within Southeast Asia and worldwide. The support originates primarily among middle and upper class urban populations, and when habitats are maintained in someone else's backyard, especially where animals such as tigers and elephants are concerned.

When it is time to be counted, charisma and size clearly offer real advantages. Four papers in this collection make contributions toward more cost-effective and rapid assessment of plants, animals, and ecosystems (forests and coral reefs) that people seem to care about most and to assessment of the costs of maintaining them. Beukema and van Noordwijk (this volume) successfully demonstrate the use of Pteridophytes as a recognizable plant taxon indicator to address a question that is highly relevant to densely populated Southeast Asia: to what extent do ecologically disturbed (but from a farmers' perspective enriched) forest systems retain some of the ecological character and function of the original natural forest habitats? Their answer is that the land use system matters a great deal regarding the potential to combine conservation and development. While this approach can effectively demonstrate how complex production systems and landscape mosaics may contribute to maintenance of forest-based biodiversity, no taxonomic group can be expected to be a good indicator for biodiversity as a whole, as taxa differ in their response to human-induced ecological change (Lawton et al., 1998). As an alternative to the taxonomic approach, Gillison and Liswanti (this volume) use plant functional types as a strategy to cope with the questions of vegetation structure and diversity in life forms. They also link their functional indicators approach to abiotic factors to inform sampling and produce a much more efficient assessment of a wider range of variation. Loreau et al. (2001) suggest that such abiotic factors "tend to be the main drivers of variations in ecosystem processes across environmental gradients".

Among the three themes of this collection, the functional role of biodiversity at the landscape level is by far the most difficult conceptually and empirically. The analysis of conflict between wildlife and people by Nyhus and Tilson (this volume) tackles this issue for two big animals (tigers and elephants) that each are icons of high global values even as they may impose costs on local human populations, who suffer property damage, personal injury, and death. Evidence presented by Nyhus and Tilson that risks to people from wildlife (and vice versa) peak in moderately disturbed systems (like agroforestry) highlights a troubling dimension of hopes for the integration of conservation and development objectives in the segregate–integrate analysis of multi-use landscapes (Van Noordwijk et al., 1997), especially since more integrated landscapes can also increase local values of other (less charismatic) types of biodiversity that can be (and often are) harvested sustainably for a range of products for local use.

All five of the papers on biodiversity in this collection attest that pragmatic approaches can produce valid conclusions that have policy relevance without attempting to measure everything, despite mind-boggling biological richness and ecological complexity. The Participatory Rapid Economic Valuation (PREV) methodology illustrated by a case study by Cannon and Surjadi (this volume) of valuation of ecotourism derived from biodiversity conservation is a particularly elegant practical demonstration that careful framing of questions to reflect specific objectives can produce useful results that economize greatly on information requirements without compromising either validity or legitimacy. Indeed, the participatory approach they used is fundamental to the usefulness, validity, and legitimacy of the results. Although the approach of comparing extreme ranges of estimates of high and low values produced a high level of confidence that ecotourism dominates other alternatives in this case—the confidence derives from use of conservative figures for the former and generous figures for the latter—this strategy will not always produce useful results without additional data and refinement. However, a general feature of this participatory approach is that it economizes on information by focusing valuation efforts on the components of greatest concern to policymakers and stakeholders. Moreover, such partial, incremental comparisons not only economize on data but also frame specific practical questions in ways that are more meaningful for policy analysis than calculation of the total value of ecosystem services (Daily et al., 2000). Estimates of total value can, however, change the public mindset (Costanza et al., 1997), even if the details do not really matter for specific decisions to be taken. Results from research by Fergus Sinclair and Laxman Joshi (see Tomich et al., 1999, p. 63, for an abstract) are encouraging regarding some scope for extrapolation of results beyond particular settings. They find evidence that suggests that farmers facing similar agroecological conditions, but in widely different locations, appear to have similar knowledge of functional aspects of biodiversity and that their apparent understanding of general patterns may be transferable across similar agroecologies. These pragmatic, participatory approaches that begin at the landscape level are not without pitfalls (what if some important dimension or threshold is overlooked?), but they are at least a way forward since it remains costly and difficult to scale-up assessments to the landscape level. And even the most cost-effective methods developed for assessment of species richness (i.e. existence) may be of limited use in assessing functional values at the landscape scale and the skills required will be agronomic, ecological, and economic rather than taxonomic.

Although the synthesis by Swift et al. (this volume; also see Loreau et al., 2001) is primarily conceptual, it too provides practical insights from better understanding of the functional roles of biodiversity that help set priorities for measurement among the bewildering range of organisms involved at the landscape level. But it is at this level where the remaining gaps in our knowledge are particularly large. Despite the efforts of Swift et al. (this volume), no functional typology at the landscape scale exists. Compared with global existence values, much less attention has been given to these functional values of biodiversity within landscapes where local people seek their livelihoods day-to-day, season-to-season, year-to-year. A dynamic view of resilience, patch dynamics and the ability to recolonize areas after an ecological disturbance is needed, but these key properties cannot be assessed easily in a survey methodology.

Anthropocentric as it may be, few would argue with the idea that being killed by a tiger or elephant would be a disservice. But beyond that, as noted by Tomich et al. in their introduction to this volume, there had been no clear consensus about the basic functions and dysfunctions of biodiversity at this scale. To remedy that deficiency, Swift et al. (this volume) developed a functional typology of the groups that support productivity, sustainability, and resilience in landscapes including agricultural uses. In addition to plants and their pollinators, the list of keystone groups by Swift et al. features parasites, micro-symbionts, decomposers, ecosystem engineers, and elemental transformers. These latter classes are the homely local 'face' of biodiversity—the millions of microbes next door and mycorrhizal fungi under our feet—and are not at all charismatic (to most people). By any measure, most of these organisms live belowground or otherwise out of sight. They are tiny and tedious to classify and count.

If one species within one of the (presumably keystone) functional groups supporting agricultural production were to become extinct—which surely must be a nearly continuous process—would that matter and how would we ever know? Swift et al. describe the basic functional biodiversity rule, why any landscape needs at least one organism in each group in order to function sustainably. But is there redundancy in having many species in a functional group? Perrings (1998) has argued that '... the main external cost of biodiversity loss lies in the reduced resilience of agroecosystems in the face of environmental and market shocks'. The main evidence on these functions comes from studies of North American grasslands, which showed biodiversity plays a role in recovery of total biomass production after drought (Tilman and Downing, 1994; Gowdy, 1997; Vandermeer et al., 1998). Swift et al. (this volume) discussed the more recent interpretations of these experiments and the need to distinguish co-evolved communities from random assemblages of species, as used in the experiments. But what does this mean in practice in the tropics? Over what spatial and temporal scale should we be concerned and, if dangerous thresholds exist, how can they be detected in time to avoid catastrophe? Loreau et al. (2001) suggest that 'the relative effects of individual species and species richness may be expected to be greatest at small-to-intermediate spatial scales ... ' but more work is needed to confirm this conclusion. If the probability of catastrophe is small, but not trivial—as may be the case for biodiversity functions at the landscape scale—then Perrings et al. (1997) point to an additional methodological challenge: conventional decision models do not work well for this class of problems.

Overall, we have very few answers for the practical questions regarding biodiversity function at the landscape scale that were put forward in the introduction to this collection by Tomich et al. From a national perspective, and putting aside existence values, potential use of unique genetic resources and ecotourism potential, there is little basis for national policymakers to place the same level of concern in degradation of biodiversity in agriculture as in, say degradation of watershed functions discussed in the next section. This lack of information on functional aspects of biodiversity in Southeast Asia—particularly information that policymakers can use—is paralleled in sub-Saharan Africa (Frank Place, pers. comm.). We simply do not know what risks there are to stabilizing functions of biodiversity compared to other pressing national concerns in developing countries in the tropics nor do we have any idea of the magnitudes of potential losses if a threshold is crossed.

In particular, the central question of the value of redundancy within functional groups remains one of the 'grand challenges of environmental science' (National Research Council, 2001, pp. 20–27; Loreau et al., 2001). And, at the landscape scale, we still do not know specific threshold effects of biodiversity loss on stability of production such that land use change that could be sustainable for a limited number of actors on a limited area would be an ecological catastrophe if everyone did the same thing; nor do we have the indicators needed to assess or predict ecosystem function at this level (Hobbs and Morton, 1999; United

Nations Development Programme et al., 2000; Harvey, 2001). For example, suppose for a moment that a perennial monoculture plantation provides watershed services that are indistinguishable from natural forest. What, if anything, would be lost (or gained) on-site from conversion of natural forest to monoculture plantation in terms of stability of the production system? Perhaps an even more important question is what effect (if any) would conversion from natural forest to a monoculture plantation have on the level and stability of production off-site on land adjacent to the monoculture plantation? Would neighbors face fewer production options because of loss of wild seed sources? ... new difficulties in managing fallows or soil nutrients? ... would they suffer more (or fewer) outbreaks of pests and diseases of crops and livestock? ... or would familiar pests and diseases be replaced by exotics? In short, should the neighbors worry? In this vein, it also is worth noting that a direct use role of 'non-charismatic' elements of the local flora may be specifically important for livelihood resilience as 'famine crops' or wild species harvested for use or sale in times of hardship due to economic or climatic fluctuations.

One obvious priority for further work is whether the risk of pest and diseases increases as biodiversity richness declines within these changing landscapes (Naylor and Ehrlich, 1997). Although not often mentioned prominently by national and regional policymakers, farmers in the humid tropics typically rank crop pests and diseases (including weeds) as their paramount resource management concern. With rare exceptions (collective action for pig hunting in Sumatra, locust control, synchrony in rice planting to reduce opportunities for rats), interventions beyond the plot/household scale seem rare.

As a preliminary set of working hypotheses on these agroecological functions at the landscape level, we offer the following nested hypothesis as a basis for further applied research:

Null hypothesis. Landscape-interactions that regulate or promote problems of pests and diseases either (a) don't matter or, if they do matter, (b) are difficult to perceive or, if perceived, (c) it is difficult to effectively organize collective action to address these problems because of the usual reasons ('free riders'; monitoring, and enforcement problems).

Weeds and other crop pests and diseases, admittedly, are not very charismatic, but this may be where resilience resides in greater biodiversity at the landscape level. There has been practical demonstration of this in Asia for significant areas, albeit for the very simple case of disease control through greater *genetic* (i.e., within species) diversity in irrigated rice achieved by planting more than one rice variety within each field and coordinating this effort through collective action at the landscape level (Zhu et al., 2000); Weitzman (2000) provided a general ecological economic framework for this phenomenon. But we have very little evidence on the effects of reduction of biodiversity at the landscape level in the much more complex upland systems, in part because the measurement problems are much greater. (However, see Vandermeer et al., 1998, for general discussion and Kiss et al., 1997 for an example from temperate agriculture.)

There is, of course, the possibility that international resources and workable means will be found to protect habitat for the charismatic elements of biodiversity (or for other global public goods, such as carbon storage), and that there is enough overlap in assemblages such that richness of the very different groups of species underpinning agroecosystem resilience may be conserved as a byproduct (Daily et al., 2000). However, despite some progress in this direction, that day seems far off. In the meantime, there would seem to be an urgent need to move beyond the pioneering stage to identify whether these environmental services merit greater recognition by policymakers and, if so, to develop and validate clear, compelling examples (possibly drawn from 'natural experiments') to demonstrate why they should care.

4. Broadening and focusing questions on watershed functions

Whenever there is a flood or drought, there is a peak in public interest in 'deforestation', 'watersheds', and 'reforestation'. In contrast to biodiversity, watershed issues have had a remarkable amount of sustained attention from policymakers, not to mention billions of dollars in public funds. But which among (a) *on-site* effects of soil erosion on productivity, (b) *off-site* effects of soil transfer on agricultural productivity and other effects, such as sedimentation of reservoirs, (c) *flooding*, (d) *seasonal water shortages*, and (e) *water pollution* from land use are of greatest concern at various scales (communities, provinces, nations)? Kramer et al. (1998, p. 2) observed that 'most analyses of watershed services have focused on soil erosion effects. Studies of other watershed services, such as streamflow stabilization, water quality and quantity effects (particularly in the case of tropical settings) have seldom been done'. Despite decades of research, it appears that science has produced surprisingly little useful information for policy questions about different watershed functions.

Policy analysis seems to be incomplete even for the topic that has received most emphasis by researchers, the plot-scale effects of erosion on agricultural productivity. Lal (1998), one of the best known researchers in the field, concluded that 'agronomic effects of erosion on crop yield have not been adequately assessed. ... A major cause of controversy and confusion about the agronomic impact of erosion is due to weak, incomplete and unreliable data on soil erosion and its impact on productivity'. Based on careful econometric analysis of data from soil samples taken intermittently since the early part of the 20th century in Indonesia (and also since the late 1930s in China), Lindert (1998) concluded that the analysis of erosion 'failed to show that it was a key source, or an accelerating source, of soil degradation in Indonesia over this half century ... Perhaps research on soil degradation should concentrate less on erosion and more on other human-induced processes, such as fertilizer, water control, and nutrient depletion'.

Policy-relevant results are even thinner regarding sedimentation and other downstream effects of soil transfer. Erosion from steep slopes and deposition in the lowlands could increase or decrease aggregate agricultural production at the watershed scale. But, even if erosion were to halt completely in Southeast Asia, it is impossible to know the likely effect on agricultural productivity. Again, Lal (1998) suggested that much remains to be done:

"... the magnitude of soil erosion for principal soils and ecoregions is also not known. The available information on the magnitude or severity of soil erosion, voluminous and often replete with rhetoric is confusing, qualitative, incomplete, and unreliable. ... The information on soil erosion is also erratic because of lack of scaling procedures. It is difficult to aggregate the data from point or field scale to landscape, watershed, ecoregional and global scales".

The model developed by Shively and Coxhead (this volume) may well represent the state-of-the-art of policy analysis of the economic effects of erosion at the landscape scale. Their stylized model (i.e. highly simplified to reveal certain key relationships more clearly) uses conventionally available data to examine erosion outcomes on upland crop productivity. Although based on an extremely narrow view of farm household decision making, the results for on-farm effects are particularly revealing. Their stylized upland farms 'choose' higher profitability (but eroding) activities, even for discount rates of only 5% on future productivity decreases on-site. In other words, the optimal rate of erosion from a private perspective (without considering the off-site effects) is almost certainly greater than zero. This, of course, is anathema to soil conservation programs seeking to 'eliminate erosion' by preaching to farmers about losses of on-site productivity. But, despite the (vastly) simplified assumptions in the model, this result does not seem unrealistic, and is consistent with the apparent need for substantial government subsidies (as usually associated with soil conservation programs worldwide), or as they demonstrate, disincentives to production of erosion-prone crops. For poor upland farmers facing difficult choices and much higher interest rates in the real world, gradually declining productivity may well appear to be a fair trade-off for more money right now.

As Shively and Coxhead recognize, the limitations in the data typically collected by soil scientists and agronomists severely restrict the *off-site* effects that can be modeled. Unfortunately, these off-site effects are the ones of greatest potential interest for environmental policy and only the simplest sort of lateral flow can be captured in the Shively–Coxhead model compared to the range of policy-relevant possibilities identified by Van Noordwijk et al (this volume). It is noteworthy that Shively and Coxhead are able to incorporate one major economic externality, the accumulation of sediment at downstream locations. However, filter effects in the uplands, which could moderate sedimentation in downstream irrigation systems, and effects on lowland agricultural productivity—which could be positive or negative—could not be modeled because of the sorts of data problems mentioned in Lal's critique quoted above. It is important to emphasize that this is not a problem of lack of measurement or an isolated botched case; rather, these problems appear to derive from application of standard soil measurement practices.

Similarly, the economic valuation of the effects of land use change on seasonal water shortages (known as base flows, low flows, or minimum flows) by Pattanayak (this volume) is constrained by availability of hydrological data. In this case, the problem is not one of technique. Instead, as Mungai et al. (this volume) argue, long-term studies (lasting decades, not seasons) may well be necessary to produce credible evidence on longer-term phenomena, such as the effects of deforestation or reforestation on base flows, a topic that will be taken up again below. Moreover, Bruijnzeel (this volume) urges research that supplements the well-established paired catchment approach with process measurements and physically based model applications to improve understanding of effects on low flows of filter elements and specific vegetation types within upper watershed landscapes.

Indeed, while many challenges also remain in economic theory and quantitative spatial methods, it appears that real progress on economic valuation of watershed functions depends on reorientation by soil scientists, hydrologists or physical geographers to measurement of the various lateral flows involved in an explicitly spatial framework. As Van Noordwijk et al. (this volume) have pointed out, spatially explicit biophysical modeling of lateral flows can be an especially enlightening way to 'read the landscape', and thereby focus measurement activity where it will count. Measurements of high within-plot soil movement but low sediment transfers to streams were discussed by Rodenburg et al. (2003). At least, for the quick processes (erosion, sedimentation, flooding, possibly water pollution) there are prospects of obtaining useful new data on effects of land use with a few seasons of well focused, directed and located observations.

The work by Ziegler et al. (this volume) epitomizes what can be accomplished through savvy scientific efforts to measure lateral flows. They provide evidence that unpaved roads produce as much sediment as agricultural land in an upper catchment in Northern Thailand, despite the fact that these roads occupy less than one-tenth of the area occupied by agriculture. Bruijnzeel (this volume) presents additional evidence of disproportionate erosion rates on (incompletely) compacted surfaces such as roads, paths, tracks, and human settlements. Further stages of compaction may lead to runoff without much soil loss, but surface flows may pick up soil as soon as they pass over soil with a higher propensity to entrainment elsewhere. Although conversion of forests to agriculture invariably is accompanied by tracks, roads, and settlements, the focus of most researchers on the former with almost complete neglect of the latter suggests an inadvertent 'misreading' of landscape processes, at least in the case of soil transport and sedimentation.

Just as with the critique of conventional wisdom and measurement practices regarding soil erosion and sedimentation, fundamental questions have been raised in the past few years about the hydrological functions of forests compared to alternative land uses. Over the past decade, numerous reviews of available evidence have concluded that deforestation has little impact on flooding (Chomitz and Kumari, 1996; Calder, 1998) and that forests (whether natural or plantation) 'use more water than most agricultural crops or grassland' (Bruijnzeel, 1990). Chomitz and Kumari (1996) summed up the emerging revisionist mood: '... the levels of the [hydrological] benefits are poorly understood, likely to be context-specific, and may often be smaller than popularly supposed'.

Based on the most recent results, Bruijnzeel (this volume) revisits these uncertainties about basic relationships between rainfall, watershed functions, deforestation, reforestation and other aspects of land use change in the humid tropics. We focus here on two elements of his comprehensive review: flooding risk and, conversely, low flow (risk and severity of water shortages). Bruijnzeel finds convincing evidence linking deforestation to increased local risks of flooding (i.e., within small catchments). But, while the possibility cannot be ruled out, he finds no comparable body of evidence linking deforestation to flooding in larger areas. Similarly, a summary of opinions expressed in an 'electronic workshop' organized by the FAO indicated no case of measurable land use impacts on peak flow (or base flow) in basins over 100 km^2 (Kiersch and Tognetti, 2002). Thus, if such impacts exist, they have yet to be clarified and measured through research. What is clear, however, is that 'truly devastating' major floods are, in Bruijnzeel's words, generally the result of a 'large and persistent field of extreme rainfall

. . . particularly when it occurs at the end of the rainy season' when soils already are saturated. This helps explain why land cover may matter least during the extreme events that produce large-scale floods.

Since extreme rainfall is the dominant factor in the worst floods—including localized flash floods (and deep landslides), as well as larger general floods—this suggests that greater attention be paid to risk assessment of settlement locations (especially expansion of settlements in floodplains) than to the often futile efforts to influence land use in upper watersheds discussed in the next section. In addition, monitoring of rainfall across catchments to provide early warning to lowland areas when rainfall exceeds dangerous thresholds could reduce risk of human tragedies that so often have made headlines in Southeast Asia. Of course, some idea of the threshold level is necessary to implement this approach.

Pioneering efforts in Thailand are involving local people in monitoring rainfall and making assessments of related risks. After recent flash floods and landslides associated with extreme rainfall patterns in some highland areas, trials of early warning functions are being incorporated into pilot local watershed service monitoring networks in sub-watershed areas of Mae Chaem, Northern Thailand, which primarily utilize data collected, analyzed, and used by local communities themselves. Since there are tensions between some upper watershed villages and their lowland counterparts that center on lowland criticism related to the impact of highland agricultural land uses on downstream watershed services, upstream villagers' efforts in support of this early warning system may help improve channels of communication and relationships with downstream communities. Villagers are also beginning to move some recent settlements in high-risk floodplain areas. Moreover, a major focus of these monitoring networks is on attention to other related issues, such as water quality (using biological indicators), seasonal stream flow (with particular interest in low flows), and soil movement in different types of agricultural fields. In this way, the broader monitoring system established under the later stages of the 'environmental issue cycle' (Tomich et al., this volume) may speed recognition of emerging problems as conditions change, possibly leading to reduced impacts and/or lower mitigation costs in a new 'issue cycle'.

Determining effects on low flows of filter elements and specific vegetation types within watershed landscapes is identified by Bruijnzeel (this volume) and Bruijnzeel et al. (2004) as the single most urgent watershed research need, because we still are unable to make real predictions for any particular area, and especially under mosaic landscape conditions common in Southeast Asia. In order to accomplish this complex task, he recommends that the traditional paired catchment approach be supplemented with spatially explicit distributed hydrological process models capable of representing complex feedback mechanisms between climate, vegetation and soils at multiple spatial scales. Refinement of the latter will also require carefully targeted systematic measurement of hydraulic characteristics under post-forest land cover types. Although it is clear that reforestation and soil conservation can reduce enhanced peak flows and stormflows associated with soil degradation, there is no well-documented case where this has produced an increase in low flows. If it turns out that land surface conditions and soil characteristics are indeed more important than tree cover per se in determining land use impacts on base flow, this would have major implications for watershed policies and programs.

5. Quest for policy levers that can influence land use: rewards or regulations?

Better understanding of the various drivers of land use and cover change may well be more important to certain regional and national policymakers than the changes in landscape structure and environmental services that result. What policies and institutional options *really* can influence the rate and pattern of land use change? Policy options of particular interest cluster under two broad categories: (1) regulations, which are the more traditional administrative approach, and (2) rewards, used as shorthand here to refer to various new ideas for environmental service incentives, which are usually positive (e.g., payments, subsidies investment in services or infrastructure), but in principle could also include 'negative rewards' (e.g., taxes, penalties, and other sanctions).

Some might guess that market-based rewards always would beat regulations in terms of efficiency of implementation and effectiveness of outcomes.

Weitzman (1974), however, showed how the choice between rewards and regulations depends on specific technical, institutional and informational circumstances and, above all, on uncertainty. On the technical side, regulation may be the better option when there are important threshold levels for damage or benefits and the system is incompletely reversible. Basically, a 'don't cross this line' rule saves on monitoring costs since enforcement effort focuses on those close to (or beyond) the threshold. Smoke pollution is one example where thresholds matter; some countries in Southeast Asia already have air quality standards that apply to smoke and particulates. Similarly, water quality (and even quantity) standards could be developed based on thresholds of damage. In principle, the threshold concept suits biodiversity functions the best. The practical problem, discussed above, is that there is not yet much empirical understanding of the stabilizing functions that really matter, so we have no idea what the thresholds are for maintaining biodiversity (or even the monitoring unit). On the institutional and informational side, uncertainty about control costs favors regulations over rewards, but uncertainty about damage has no effect on the choice of instrument from an economic perspective (Helfand, 1999). Although not insurmountable in theory (say through auctions), finding an optimal reward could be difficult in practice, particularly when there are many potential polluters or providers of environmental services, each with very different cost structures. So economic theory would tend to point toward regulation rather than reward for the environmental services discussed in this collection and also for upland farmers who are potential polluters/providers of these services in Southeast Asia.

In reality, however, regulations aimed at forest protection have had mixed success, at best, and land use planning has even less impact on the ground in Southeast Asia. When government regulations have attempted to impose '*protected*' status on areas of land or water, historical, cultural, or de facto established rights of local people (who are often ethnic minorities in mountain areas where most terrestrial protected areas have been declared) typically were not adequately respected or even recognized, nor were these people compensated for foregone resource use and loss of development opportunities that protection would entail. As a result, the regulatory approach to environmental policy often has been ineffective because of high transactions costs resulting from incentive incompatibility and limited administrative capacity, or even counterproductive, because it perverts and destroys local resource management incentives. Worse yet, government regulations aimed at resource protection have mandated expulsion of people from 'protected areas', depriving them of their land and livelihoods and forcing some into poverty and further resource degradation elsewhere.

Yet, there is cause for some hope. Conservation can produce local benefits as well. In substantial parts of the humid tropics, local forms of land use have emerged that allow people to make a living while protecting environmental resources. The resulting levels of environmental services are below those of (perceived) 'pristine' nature, but are superior to many other agricultural development options from an environmental perspective. The 'agroforests' of Southeast Asia (with counterparts across the humid tropics) are a prime example of how 'domesticated forests' can provide food, timber and income, while harboring a substantial share of the original forest biodiversity, which often lacks adequate protection elsewhere. Depending on commodity prices, investment opportunities and government policies, however, the small-scale managers of the agroforests can be (and are) induced to replace these systems by monocultural plots of oil palm, rubber or other crops. Who can blame them for doing so, if the outside world has not found ways to express their appreciation for the environmental qualities of the agroforests in a way that is meaningful for the farmers? Similar issues relate to community-based management of mosaic agroforestry landscapes in mountainous areas of mainland Southeast Asia whose centuries-old long-rotation forest fallow land use systems have only relatively recently come into question, and are now under heavy pressure to convert to intensive permanent commercial field crop production on sloping lands.

Workable means for effective recognition and rewards for environmental services are receiving increasing attention (Fig. 1; also see Johnson et al., 2001; Pagiola et al., 2002; Powell et al., 2002; Scherr et al., 2002). Measures to create appropriate rewards could emerge as an important part of the answer to the practical and ethical dilemma of how to reconcile broad environmental objectives and local livelihood

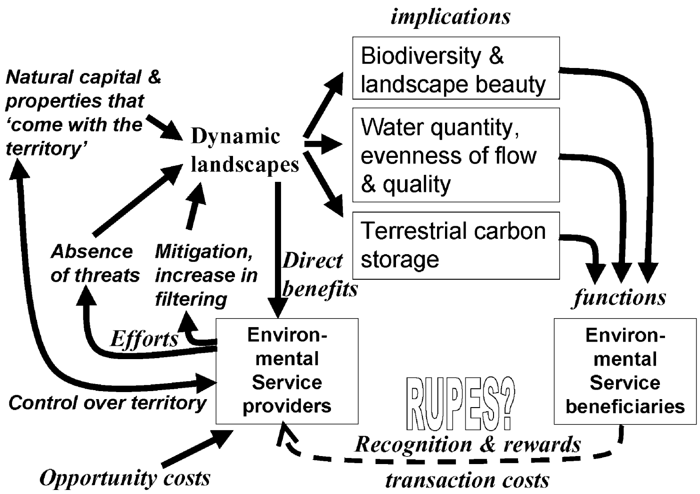

Fig. 1. Examples of the landscape-level implications that are perceived as environmental service functions and the type of efforts and activities that can qualify actors as 'environmental service providers': absence of threats and mitigation or increase in filter functions.

imperatives—and may even play a complementary role with the regulatory approach to environmental protection.

Putting such reward systems into practice often raises a host of cultural issues and institutional questions. Consider the case of the Mae Taeng watershed in Northern Thailand, where water yields had declined for two decades. Although it was not possible to definitively identify the complex causal factors underlying this trend, it was clear that competition for water between upstream, agricultural uses and downstream, urban uses was increasing (Vincent et al., 1995). The research team observed that 'it may be cheaper ... to allow farmers from the Irrigation Project area to sell their water to users in the city' but 'buying water from farmers would require an institutional revolution in Thailand ... ' (Vincent et al., 1995, pp. vi–vii). To date, such a market-based approach has not been tried, and it remains to be seen whether it would be possible to create and manage mechanisms for compensating people for foregone livelihood opportunities in favor of environmental services to other groups. Where there is only limited potential for influencing the total amount of water available from rainfall minus evapotranspiration by natural vegetation, changes in the use of water (for indirectly supporting dry season evapotranspiration in irrigation schemes) provide scope for increasing domestic or industrial water use. Selling water that is not used for irrigation almost invariably raises questions of ownership and rights to sell, however, that are deeply rooted in society and are not easy to answer.

In the cause–effect relations that underlie the generation (or degradation) of environmental services, we can generally distinguish three key elements (Fig. 2): natural capital (including rainfall and inherent richness of flora and fauna), a 'guardianship' role of preventing destruction of the natural capital that itself largely depends on social capital, and an active management or stewardship role that is part of the human

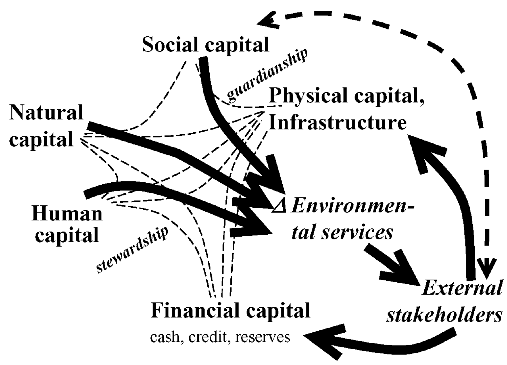

Fig. 2. Relationship between the five types of capital (Carney, 1998), changes in environmental services, and the possible rewards provided by external stakeholders.

capital, but which also draws on social capital for adaptation and replication across generations. Reward mechanisms can take the form of direct payments (financial capital), support for infrastructure (physical capital), rights to use natural capital (for consumption or investment in other forms of capital) that typically are socially mediated and controlled (hence based on social capital), indirectly through say investments in social or human capital (support for clinics or schools conditioned on supply of environmental services) or some mix of these options. The distinctions among these five types of capital matter not only because of the distinct functions above but also because of their different characteristics (e.g., regarding time frame for reversibility of investment decisions) and because of imperfect scope for substitution between them (e.g., money cannot buy trust). So, for example, a simple market-based approach relying on financial payments and competitive price formation may prove ineffective or unsustainable if complementary investments to build trust across spatial and institutional scales are ignored. Moreover, since the costs of different options for implementation rest on existing endowments of these five types of capital, the optimal mix of reward mechanisms (in the sense of least transaction costs to sustain a particular level of environmental service) likely will vary greatly across countries and even between neighboring communities.

A comparative action research approach based on pilot projects, such as the Rewarding the Upland Poor for Environmental Services (RUPES) project in Southeast Asia (Fig. 1; http://www.worldagroforestrycentre.org/sea/Networks/RUPES/), would seem to be a timely and essential step toward filling gaps in our knowledge of effective implementation of reward-based environmental policy levers.

6. Conclusion: toward nested levels of understanding, governance and equity

Tomich et al. introduced this volume with a set of questions about problems at different spatial scales and different sequential stages of an environmental issue cycle. Issues of problem recognition, perception, and requirements for measurement and scaling vary through that cycle and cut across this collection. Indeed, identifying the scale of analysis—and of intervention—depends on the specifics of a problem and often varies through the cycle.This collection has focused on three environmental issues as the ‘meso’-scale, that is they have important lateral flows (Van Noordwijk et al., this volume), but they are not global. Perhaps because fires and smoke are readily detectable with remote sensors, the smoke management issue is the most fully developed regarding empirical scaling. For watersheds, too, there has been progress on identifying the scale of specific dimensions of those functions (Bruijnzeel, this volume; Kiersch and Tognetti, 2002). The scale concept is least developed in the case of landscape-level biodiversity functions and indeed for ecology more broadly (Schneider, 2001). This certainly is related to our limited understanding of those functions. However, even for biodiversity functions, there are early suggestions of the scale of the services. As mentioned above, Loreau et al. (2001) suggest that these effects may be greatest at ‘small-to-intermediate spatial scales’. Similarly too for watersheds, at least for the cases of erosion, sedimentation, and flooding (but perhaps not for base flow or water quality?), the primary effects of land cover change occur within smaller catchments. And even in the case of the ‘regional smoke problem’, both the costs and the most promising interventions were shown to be essentially local. So for many (but not all) of our themes, impacts and actions at intermediate-to-local ‘meso-scales’ appear likely to play a critical role in efforts to effectively address these environmental issues, which are linked to land use and land use change.

For global–local conflicts regarding the environment—for example, the case of human wildlife conflict presented by Nyhus and Tilson (this volume)—the human ‘stakeholders’ with conflicting interests typically never meet. However, for landscape-level environmental issues, political and social activity and overt conflict focused on land use and cover change may be an important indicator of the existence of significant environmental issues at the landscape level, as well as a reflection of needs to strengthen resource management capacities at intermediate-to-local levels of governance.

In this sense, trends toward more decentralized and democratic governance systems in a number of Southeast Asian countries may bode well for improved capacity to manage these environmental services. However, a recent synthesis of environmental governance case study findings in mainland Southeast Asia by Dupar and Badenoch (2002) indicates there are still many questions about the appropriate scope of powers located at different levels of governance hierarchies, as well as about incentives, accountability processes and fiscal arrangements of intermediate-to-local level institutions appropriate for managing natural resources. Further efforts to improve our understanding of biophysical processes and functions at different nested spatial scales of analysis could help in formulating appropriate mandates for different jurisdictional levels, in identifying spatial domains that provide and benefit from various types of environmental services, and in valuing and monitoring environmental service flows. This could help strengthen foundations for further refinement and testing of environmental service reward mechanisms.

But this multi-level approach implies that in addition to the gap between researchers and national policymakers, there also is a new urgency toward reaching across the gaps that often separate analysts, policymakers, specialized government agencies, 'civil society', business interests, and other major stakeholder groups at key spatial scales of resource governance. Because of the likelihood of conflicting interests, it is naïve to expect that research alone will be sufficient to produce and implement better public policy. Social and political mechanisms will be needed to support negotiations to address these conflicts (Van Noordwijk et al., 2001; Wollenberg et al., 2001) within and among appropriate units of governance. In the likely case there are winners and losers, the challenge is to strengthen or create mechanisms for negotiation support and conflict management—between neighboring communities; upstream and downstream populations; local, national, or perhaps even global concerns–that promote social justice as well as environmental benefits. Thus, environmental issues and opportunities may be even more complex than commonly perceived. But recognition of the scope of this complexity based on more robust understanding of underlying biophysical processes and human behavior also helps expand the range of policy options for supporting the sustainable provision of environmental services and provides a rich set of conditions for more systematic strategic testing of key concepts and mechanisms within the context of continually emerging and evolving environmental issue cycles.

Acknowledgements

In addition to the authors of the other articles in this collection, we also wish to acknowledge the insights and ideas we received from policymakers and scientists who participated in a Southeast Asian regional workshop on 'Environmental Services and Land Use Change: Bridging the Gap between Policy and Research in Southeast Asia' held in Chiang Mai, Thailand, from 31 May to 2 June 1999 and a global workshop on 'Bringing the Landscape into Focus: Developing a Conceptual Framework and Identifying Methods for ASB Work at the Landscape Level', held in Chiang Mai, 12–13 November 2001 and organized in collaboration with the Alternatives to Slash-and-Burn (ASB) thematic working group on Sustainable Land Use Mosaics, led by Stephan Weise of the International Institute for Tropical Agriculture (IITA), and helpful comments and suggestions from our colleagues Marian de los Angeles, Richard Coe, Brent Swallow, and Bruno Verbist.

References

Applegate, G.B., Chokkalingam, U., Suyanto, S., 2001. The underlying causes and impacts of fires in Southeast Asia. Final Report. Center for International Forestry Research, International Centre for Research in Agroforestry, United States Forest Service, Bogor, Indonesia, p. 58.

Brookfield, H., Potter, L., Byron, Y., 1995. In Place of the Forest: Environmental and Socio-economic Transformation in Borneo and the Eastern Malay Peninsula. United Nations University Press, Tokyo.

Bruijnzeel, L.A., 1990. Hydrology of moist tropical forests and effects of conversion: a state of knowledge review. Humid Tropics Programme. UNESCO International Hydrological Programme.

Bruijnzeel, L.A., Bonell, M., Gilmour, D.A., Lamb, D., 2004. Forests–water–people in the humid tropics: an emerging view. In: Bonell, M., Bruijnzeel, L.A., Forests, Water and People in the Humid Tropics. Cambridge University Press, Cambridge, UK.

Calder, I.R., 1998. Water-resource and land-use issues. SWIM Paper 3. International Water Management Institute, Colombo, Sri Lanka.

Carney, D., 1998. Implementing the sustainable rural livelihoods approach, sustainable rural livelihoods—what contribution can we make? Department for International Development, London, UK, pp. 3–23.

Chomitz, K., Kumari, K. 1996. The domestic benefits of tropical forests: a critical review emphasizing hydrological functions. Policy Research Working Paper 1601. Policy Research Department, The World Bank, Washington, DC.

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

Daily, G.C., Soderqvist, T., Aniyar, S., Arrow, K., Dasgupta, P., Erlich, P.R., Folke, C., Jansson, A., Jansson, B.-O., Kautsky, N., Levin, S., Lubchenco, J., Maler, K.-G., Simpson, D., Starrett, D., Tilman, D., Walker, B., 2000. The value of nature and the nature of value. Science 289, 395–396.

Dennis, R., Mayer, J., Applegate, G.B., Suyanto, S., Stolle, F., Kurniawan, I., Colfer, C.J.P., Permana, R.P., Ruchait, Y., Maus, P., Lawkowski, H., Tomich, T.P., 2004. Fire, people, and pixels: linking remote sensing and social science to understand underlying causes and impacts of fires in Indonesia, (under review).

Dupar, M., Badenoch, N., 2002. Environment, Livelihoods, and Local Institutions: Decentralization in Mainland Southeast Asia. World Resources Institute, Washington, DC.

EEPSEA (Economy and Environment Program for SE Asia), 1993–1998. The Economy and Environment Program for Southeast Asia, Singapore.

EEPSEA (Economy and Environment Program for SE Asia) and WWF (World Wide Fund for Nature), 1998. New estimates place damage from Indonesia's 1997 fires at $ 4.4 billion. Press release 29 May 1998.

Glover, D., Jessup, T., 1998. The Indonesian fires and haze of 1997: the economic toll. http://www.idrc.org.sg/eepsea/

Glover, D., Jessup, T., 1999. Indonesia's Fires and Haze: The Cost of Catastrophe. Institute for Southeast Asian Studies, Singapore, and International Development Research Centre, Canada, Singapore.

Gowdy, J.M., 1997. The value of biodiversity: markets, society, and ecosystems. Land Econ. 73 (1), 25–41.

Harvey, J., 2001. The natural economy. Nature 413, 463.

Helfand, G.E., 1999. Standards versus taxes in pollution control. In: Van den Bergh, J.C.J.M. (Ed.), Handbook of Environmental and Resource Economics. Edward Elgar, Cheltenham, UK, Chapter 15, pp. 223–234.

Hobbs, R.J., Morton, S.R., 1999. Moving from descriptive to predictive ecology. Agrofor. Syst. 45, 43–55.

Johnson, N., White, A., Perrot-Maitre, D., 2001. Developing Markets for Water Services from Forests: Issues and Lessons for Innovators. Forest Trends, World Resources Institute, and the Katoomba Group, Washington, DC, 19 pp.

Kiersch, B., Tognetti, S., 2002. Land-water linkages in rural watersheds: results from the FAO electronic workshop. Land Use Water Resour. Res. 2, 1.1–1.6.

Kiss, J., Penksza, K., Toth, F., Kadar, F., 1997. Evaluation of fields and field margins in nature production capacity with special regard to plant protection. Agric. Ecosyst. Environ. 63, 227–232.

Kramer, R.A., Pattanayak, S., Priyanto, A., 1998. Watershed protection benefits and the Ruteng nature recreation area in Flores, Indonesia. Nicholas School of the Environment, Duke University, Durham, NC.

Lal, R., 1998. Agronomic consequences of soil erosion. In: Penning de Vries, F.W.T., Agus, F., Kerr, J. (Eds.). Soil Erosion at Multiple Scales. CAB International, Wallingford.

Lawton, J.H., Bignell, D.E., Bolton, B., Bloemers, G.F., Eggleton, P., Hammond, P.M., Hodda, M., Holt, R.D., Larsen, T.B., Mawdsley, N.A., Stork, N.E., Srivastava, D.S., Watt, A.D., 1998. Biodiversity inventories, indicator taxa and effects of habitat modification in tropical forest. Nature 391, 72–76.

Lindert, P., 1998. Shifting Ground: The Changing Agricultural Soils of China and Indonesia. University of California, Davis, CA.

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J.P., Hector, A., Hooper, D.U., Huston, M.A., Raffaelli, D., Schmid, B., Tilman, D., Wardle, D.A., 2001. Biodiversity and ecosystem functioning: current knowledge and future challenges. Science 294, 804–808.

National Research Council, 2001. Grand Challenges in Environmental Sciences. National Academy Press, Washington, DC.

Murdiyarso, D., Widodo, M., Suyamto, D., 2002. Fire risks in forest carbon projects in Indonesia. Sci. China, Ser. C 45, 65–74.

Naylor, R.L., Ehrlich, P.R., 1997. Natural pest control services and agriculture. In: Daily, G.C. (Ed.), Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, DC, Chapter 9, pp. 151–174.

Pagiola, S., Bishop, J., Landell-Mills, N. (Eds.), 2002. Selling Forest Environmental Services: Market-based Mechanisms for Conservation and Development. Earthscan Publications, London.

Perrings, C., Maler, K.-G., Folke, C., Holling, C.S., Jansson, B.-O. (Eds.), 1997. Unanswered Questions. Biodiversity Loss: Economic and Ecological Issues. Cambridge University Press, Cambridge, UK.

Perrings, C., 1998. The economics of biodiversity loss and agricultural development in low income countries. Paper Presented at the American Association of Agricultural Economics International Conference on 'Agricultural Intensification, Economic Development, and the Environment', Salt Lake City, USA, 31 July–1 August.

Powell, I., White, A., Landell-Mills, N., 2002. Developing Markets for the Ecosystem Services of Forests. Forest Trends, Washington, DC, 12 pp.

Reinhardt, T., Ottmar, R., Castilla, C., 2001. Smoke impacts from agricultural burning in a rural Brazilian town. J. Air Waste Manage. Assoc. 51, 443–450.

Rodenburg, J., Stein, A., Van Noordwijk, M., Ketterings, Q.M., 2003. Spatial variability of soil pH and phosphorus in relation to soil run-off following slash and-burn land clearing in Sumatra, Indonesia. Soil Till. Res. 71, 1–14.

Scherr, S., White, A., Kaimowitz, D., 2002. Making markets work for forest communities. Policy Brief. Forest Trends, Washington, DC and Center for International Forestry Research, Bogor, Indonesia.

Schneider, D.C., 2001. The rise of the concept of scale in ecology. BioScience 51 (7), 545–553.

Stolle, F., Tomich, T.P., 1999. The 1997–1998 fire event in Indonesia. Nat. Res. 35, 22–30.

Suyanto, S., Applegate, G., Taconi, L. 2002. Community-based fire management, land tenure and conflict. In: Moore, P., et al. (Eds.), Communities in Flames, Proceedings of an International Conference on Community Involvement in Fire Management. Food and Agriculture Organization of the United Nations, Regional Office for Asia and the Pacific, Bangkok, Thailand, pp. 27–32.

Tilman, D., Downing, J.A., 1994. Biodiversity and stability in grasslands. Nature 367, 363–365.

Tomich, T.P., Fagi, A.M., De Foresta, H., Michon, G., Murdiyarso, D., Stolle, F., Van Noordwijk, M., 1998. Indonesia's fires: smoke as a problem, smoke as a symptom. Agrofor. Today 10 (1), 4–7.

Tomich, T.P., Thomas, D.E., Van Noordwijk, M. (Eds.), 1999. Research Abstracts and Key Policy Questions. Environmental Services and Land Use Change: Bridging the Gap Between Policy and Research in Southeast Asia. ASB-Indonesia Report Number 10. ASB-Indonesia and ICRAF SE Asia, Bogor.

Tomich, T.P., Lewis, J. (Eds.), 2002. Reducing Smoke Pollution from Tropical Fires. ASB Policy brief No. 4. Alternatives to Slash-and-Burn Systemwide Programme. World Agroforestry Centre, Nairobi, Kenya. http://www.asb.cgiar.org.

United Nations Development Programme (UNDP), United Nations Environment Programme (UNEP), World Bank, World Resources Institute (WRI), 2000. World Resources 2000–2001: People and Ecosystems: The Fraying Web of Life. World Resources Institute, Washington, DC.

Vandermeer, J., Van Noordwijk, M., Anderson, J., Ong, C., Perfecto, I., 1998. Global change and multi-species agroecosystems: concepts and issues. Agric. Ecosyst. Environ. 67, 1–22.

Van Noordwijk, M., Tomich, T.P., De Foresta, H., Michon, G., 1997. To segregate or to integrate? The question of balance between production and biodiversity conservation in complex agroforestry systems. Agrofor. Today 9 (1), 6–9.

Van Noordwijk, M., Tomich, T.P., Verbist, B., 2001. Negotiation Support Models for Integrated Natural Resource Management in Tropical Forest Margins. Conservation Ecology 5 (2). http://www.consecol.org/vol5/iss2/art21.

Vayda, A.P., 1998. Findings on causes of the 1997–1998 Indonesian forest fires: problems and possibilities. Preliminary Research Report. WWF-Indonesia Program, Jakarta.

Vincent, J., Kaosa-ard, M., Worachai, L. 1995. The Economics of Watershed Management: A Case Study of the Mae Taeng. Thailand Development Research Institute, Bangkok.

Weitzman, M., 1974. Prices versus quantities. Rev. Econ. Stud. 41, 477–491.

Weitzman, M., 2000. Economic profitability versus ecological entropy. Quart. J. Econ. 115, 237–263.

Wollenberg, E., Edmunds, D., Anderson, J. (Eds.), 2001. Accommodating multiple interests in local forest management. Int. J. Agric. Resour., Govern. Ecol. 1 (3–4) (special issue).

Wuethrich, B., 2000. Combined insults spell trouble for rainforests. Science 289, 35–37.

Zhu, Y., Chen, H., Fan, J., Wang, Y., Li, Y., Chen, J., Fan, J., Yang, S., Hu, L., Leung, H., Mew, T., Teng, P., Wang, Z., Mundt, C., 2000. Genetic diversity and disease control in rice. Nature 406, 718–722.