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The Role of Forest Biodiversity in the Sustainable Use of Ecosystem Goods and Services in Agro-Forestry, Fisheries, and Forestry

Proceedings of International Symposium
for the Convention on Biological Diversity

April 26 - 28, 2010
Tokyo, Japan

Forestry and Forest Products Research Institute

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Edited by Toru Koizumi, Kimiko Okabe, Ian Thompson, Ken Sugimura, Takeshi
Toma, and Kazuyuki Fujita



Forestry and Forest Products Research Institute

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Preface

This proceedings is a compilation of papers presented at an International Symposium on "The role of forest biodiversity in the sustainable use of ecosystem goods and services in agro-forestry, fishery, and forestry". All papers have been peer reviewed anonymously by the qualified scientists. The symposium was organized by the Forestry and Forest Products Research Institute (FFPRI) and the Environmental Research Institute, Waseda University and held between 26 and 28 April 2010 in the Azusa Ono Memorial Hall of Waseda University, Tokyo, Japan. The symposium was sponsored by the OECD Co-operative Research Programme (www.oecd.org/agriculture/crp) and an activity of the FFPRI's special research project. In total, more than 170 participants from various organizations attended the symposium.

This symposium has been organized for providing further information to the 10th Conference of the Parties to the Convention on Biodiversity (CBD/COP10) which will be held in Nagoya, Japan in October, 2010. One of the objectives of CBD is to harmonize the conservation of bio-diversity with the sustainable use of the components of biodiversity. Recently, it has been recognized that conserving forest ecosystem leads to serve various goods and services to forests and their surrounding environments. In the symposium, we have focused on forest biodiversity and its importance for agriculture, forestry and fishery so that we can increase our understanding on values of forest biodiversity as ecosystem services which are vital to our society.

We, scientists, have been accumulating knowledge and information, which are relevant for conserving biodiversity for decades. In FFPRI, we have had a special research project for contributing the success of CBD/COP10. FFPRI and its partner institutions have carried out various studies on conservation of biodiversity in forest ecosystems intensively for last decades. It is a right time to exchange our knowledge and deliver it to the society. Then, we have invited noted scientists who are all on the front line of the world level as the speakers or the organizers of this symposium. During the symposium, the latest research information are delivered by the top scientists and discussed by the scientists and also non-scientists with various backgrounds.

I hope that this proceedings as the outputs of the symposium becomes instructive scientific messages for the success of CBD/COP10.

I greatly thank to all participants of the symposium, especially to two chairs of the symposium Drs. Ian Thompson and Kimiko Okabe. I also thank to following co-organizing organizations and institutions: Secretary of CBD (Convention on Biological Diversity), CFS (The Canadian Forest Service), Tohoku University Ecosystem Adaptability GCOE, Nagoya City University, The National Institute for Environmental Studies, METLA (The Finnish Forest Research Institute), and CIFOR (Center for International Forestry Research).

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President,
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Ecosystem services provided by forest biodiversity

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Abstract

Biological diversity does not always play an important role for all kinds of ecosystem services. Some ecosystem services are not strongly associated with biological diversity, or rather there is a tradeoff between biological diversity and a specific ecosystem service. For decision making under forest management, it is necessary to have clearer vision of the relationships between ecosystem services and biodiversity. Many experimental studies have been conducted to elucidate the relationships between biodiversity and ecosystem functions for grassland or freshwater microcosms, but very limited number of studies have been made in forested ecosystems. Furthermore, although many functions and ecosystem services have been shown to be related to biodiversity by these experimental studies, most have not dealt with very high levels (i.e., species richness or interactions) of biodiversity. In general, 1) simple ecosystems composed of a few number species (plants or trees) tend to be chosen for maximizing the provisioning services, and 2) the effects of biodiversity on many regulating services are not clear. However, regulating services like biological control or pollination are expected to be strongly related to biodiversity, and several trials to value these services have been done recently. Most of the cultural services are strongly associated with biodiversity, though they are not recognized nor highly evaluated by people. Even among the pro-environmental services, there are some tradeoffs, such as between carbon sequestration and biodiversity. In terms of provisioning and regulating services, the composition and/or proportion of components in the ecosystem are the main concern. Appropriate compositional or functional diversity may lead to a complete functioning ecosystem and thus maintenance of various ecosystem services. For cultural services, however, the species unique to a particular ecosystem and locality have important roles.

Introduction

Since Convention of Biodiversity (CBD) was launched, the importance of biodiversity has been emphasized from many perspectives. However, the value or importance of biodiversity is not yet adequately recognized by the general public yet. For forest management, application or responses to

promote conservation and sustainable use of biodiversity are also urgently needed.

The concept of ecosystem services, which was best expressed in the Millennium Ecosystem Assessment (2005), highlighted the role of ecosystems for human well-being. In this report, four categories of ecosystem services were proposed, with the list of various services under each category. However, it should be noted that all these ecosystem services are not equally associated with biodiversity (Nakashizuka 2005, Dobson et al. 2006). Biodiversity is typically important and even a small loss of biodiversity may affect the performance of some ecosystem services, though not for other services. In this paper, I will review the relationship between biodiversity and ecosystem services referring to the recent papers.

Supporting services

Supporting services are the services that support other services as the background functions of the ecosystem, such as primary production, nutrient cycling, soil formation, and so on.

Since the époque making papers like Tilman & Downing (1994) and Naeem et al. (1994), hundreds of scientific papers have been published on the role of biodiversity on ecosystem function. These studies include various trophic levels in both manipulations of biodiversity and their effects. Several experiments dealt with the effects at the ecosystem level. A meta-analysis of these papers suggests that productivity (not only primary productivity but also those at other trophic levels), nutrient cycling, erosion control, and ecosystem stability are all affected by biodiversity to certain extent (Balvanera et al. 2006). The facts that ecosystem functions and supporting services are fundamentally enhanced by biological diversity of the ecosystem have also been reported by other authors (Loreau et al. 2002, Scherer-Lorenzen et al. 2005).

These experimental studies, however, explain little about the importance of rare and/or endemic species. These experiments were to elucidate the importance of composition or diversity of components of the ecosystem on functions or services of the ecosystem. Furthermore, the number of species used in the experiments were not large, mostly less than 30 species. Thus, these studies did not succeed in explaining the role of large number of species like

those in tropical rain forests, yet.

Provisioning services

Provisioning services include the supply from ecosystems, such as food, water, fuel, fiber, chemicals, genetic and ornamental resources, and so on. For the effective supply of single resources, biodiversity does not always play an important role. For instance, monoculture or plantation of a single tree species are effective for maximizing timber provision.

Medicinal organisms or genetic resources are typical provisioning services in which biological diversity is important. In addition, most non-timber forest products (NTFPs) are goods resulting from biodiversity. While forests support the lives of the local inhabitants, they, in turn, sometimes help to conserve biodiversity by their traditional and ecological management (Kusters & Belcher 2004). Some inhabitants of tropical regions earn their livelihoods from non-timber forest products such as rattan or aromatic woods (Kusters & Belcher 2004). These species are sometimes very rare and unique to the regions. Sometimes those resources may decline because of over-exploitation owing to high commercial values placed on these products. Therefore, it is difficult to say that NTFPs always act to promote sustainable use of biodiversity.

Biodiversity is important in cases where humans require various kinds of forest products, but provisioning services are not really associated with biodiversity when we require large amounts of a single resource. The use of NTFPs sometimes helps to keep forest ecosystem sustainable, but locally unique and commercially important resources may be at risk for possible overuse.

Regulating services

Regulation services include climate regulation, flood control, detoxification, disease control, soil formation, etc. Since the performance of some of these ecosystem services, is roughly proportional to the biomass or productivity of ecosystem, the role of biodiversity may not be significant.

The function of biological control of pests is related to biodiversity (Wilby & Thomas 2002, Dwyer et al. 2004). The abundance and species richness of natural enemies sometimes play a key role in controlling outbreaks of pests (Kean et al. 2003). Pollination is another regulating service that is associated with biodiversity. Klein et al. (2007) elucidated that many agricultural crop species are deeply dependent on animal pollinators. Ricketts et al (2004) suggested that a coffee farm will suffer economic loss if the neighboring forests are removed, because these forests are inhabited by the pollinators of coffee trees.

Thus, biodiversity is not always important for all regulating services. Biodiversity is effective for

biological control of pests and pollination. However, these services have been under-evaluated and studies to value these services started only recently.

Cultural services

Cultural services include spiritual, recreational, aesthetic, educational, and symbolic services. Bio-mimicry and eco-tourism are also involved in this category. Cultural services are closely associated to both biodiversity and habitat functions.

With respect to recreation, people sometimes seek a variety of landscapes, forest structures, and compositions, all of which are components of biological diversity. Scientific and educational information are also important components of cultural services associated with biodiversity.

Biological diversity also contributes to historical, and religious aspects of most local cultures. The names of colors are one example of the cultural matters deeply influenced by biological diversity. Nagasaki (2001) reported that the Japanese people have 225 named traditional colors, attesting to their delicate sense of color. Of these, 146 colors are named after living organisms, including 120 plants, and 83 trees (Nakashizuka, 2004). This suggests that Japanese culture has developed in the context of a certain dependence on biological diversity. In another example, some local communities in Sarawak include stylized animals and plants in their traditional designs, which are associated with their collective identity as communities.

Some of these services are strongly associated with endemic species or with local biodiversity. For instance, local residents on Borneo regard some endangered species of hornbill (Bucerotinae) as symbolic. Similarly, every Japanese prefecture designates flowers and trees that symbolize the prefecture (Nakashizuka 2004).

Thus, biodiversity is important for most of cultural services, though the universal valuation of such services is difficult. The value of this type of service is sometimes strongly dependent on social background and/or history. Thus, many of the cultural services depending on biodiversity are not socially recognized or not economically well-evaluated.

Characteristics of the ecosystem services

As discussed above, ecosystem services include multi-dimensional aspects, and the necessity for biodiversity has been evaluated variously among ecosystem services. At the same time, however, some ecosystem services of a forest may conflict with others. The Millennium Ecosystem Assessment (2005) suggested that most ecosystem services have declined, while services such as the provision of foods, livestock, and aquaculture have become increasingly used over the past 50 years. These trends suggest that humans have increased provisioning services for

foods as a trade-off with other services. Furthermore, other ecosystem services may also be subject to tradeoffs with biodiversity. For instance, a forest with high carbon sequestration is not necessarily a forest with high biodiversity (Totten et al. 2003). Similarly, this can also apply to several production functions.

In general, the ecosystem services related to biodiversity tend to be evaluated relatively low compared to most other services. They have strong uncertainty and the effects on human life are sometimes vague. Also, the cultural services are not universal since it depends on social and historical background but this does not mean that such services are not important for human well-being.

Conclusions

Biodiversity is effective and/or important for some provisioning and regulating services, and most cultural services. Diversity of ecosystem components may enhance most ecosystem functions and services, although the extent may vary. On the other hand, biodiversity relating to local uniqueness are more important in cultural services. Ecosystem services to increase food provision have been enhanced by sacrificing other ecosystem services strongly related to biodiversity in recent decades. Such tradeoffs exist between ecosystem services and between ecosystem services and biodiversity. The services associated with biodiversity tend not to be evaluated very highly despite their importance to human well-being. Studies to evaluate the ecosystem biodiversity-related services are urgently necessary to improve their proper valuation.

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A synthesis on the biodiversity-resilience relationship in forest ecosystems

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Abstract

Resilience is the capacity of an ecosystem to return to a former state after a disturbance sufficiently large to alter the system in some way (e.g., fire). Ecosystem resilience is conferred at multiple scales by the biological diversity within and among genes and species within ecosystems, and among ecosystems across landscapes in forests. Genetic diversity underpins the capacity of species to adapt to changes. Functional diversity and redundancy within systems is important to maintaining resilience to change. Most studies support the concept that more diverse forests are more productive, more resistant (hence more stable), and more resilient to change than simple mono-typic stands. Ecosystem invasion is reduced by high biodiversity but paradoxes exist depending on scale. Biodiversity serves to enhance the total carbon that can be sequestered in a forest system and the production of other goods and services. Climate change will alter all forests systems, but effects will be especially negative where forests become drier, especially for many tropical forest areas. Recommendations are provided to maintain forest resilience under climate change.

Keywords resilience, biodiversity, forest productivity, forest management, climate change

Introduction

Humans are having long-term cumulative impacts on the Earth's ecosystems through a range of consumptive, exploitive, and indirect mechanisms, to the extent of now influencing the global climate (IPCC 2007). This paper describes the concept of ecosystem resilience in forests and its relationship to biodiversity, with particular reference to climate change. Emissions from deforestation and degradation remain a significant (ca. 12%) source of annual greenhouse gas emissions into the atmosphere (van der Werf et al. 2009), and therefore the conservation, appropriate management and restoration

of forests can make a significant contribution to climate change mitigation. Under severe drying conditions, forests lose resilience and may be replaced by savannahs or grasslands, while under increased temperature, open taiga can be replaced by closed boreal forests (assuming sufficient moisture) (e.g., Price and Scott 2006). Maintaining or restoring forest resilience is often cited as a necessary adaptation to climate change (e.g., Millar et al. 2007, Chapin et al. 2007) to ensure that forest goods and services persist.

Definitions

We define *resilience* as the capacity of an ecosystem (forest type) to return to the pre-condition state following perturbation, including maintaining its essential characteristic taxonomic composition, structures, and functions (Holling 1973, Peterson et al. 1998, Walker et al. 2004). Resilience is an emergent property of ecosystems that is conferred at multiple scales by genes, species, and processes within the system (Gunderson 2000, Drever et al. 2006).

Resistance is the capacity of the ecosystem to absorb disturbances and remain largely unchanged. Forests are resistant and so change little within bounds (i.e., are stable) as a result of non-catastrophic disturbances, such as chronic herbivory, minor blowdown, or canopy gaps created by the death of individual trees or small groups of trees. Forests may also be resistant to certain environmental changes such as weather patterns over time owing to redundancy at various levels among functional species. Most well-developed forests, especially primary forests, are resilient and resistant to changes (e.g., Holling 1973, Drever et al. 2006). *Stability* reflects the capacity of an ecosystem to remain approximately in the same state within bounds, that is, the capacity to maintain a dynamic equilibrium in time while resisting change. A stable system persists when it has the capacity to absorb disturbances and remain largely unchanged over long periods of time.

Functional groups are assemblages of species performing similar functions within an ecosystem (e.g., pollination or decomposition), hence providing some redundancy (e.g., see Hooper et al. 2002). Functional diversity (i.e., number of functions) is not necessarily correlated with species richness (Diaz and Cabido 2001, Hooper et al. 2005).

Several studies have shown that resilience in ecosystems is related to the biological diversity in the system and the capacity it confers to maintain ecosystem processes (Walker 1995, Peterson et al. 1998, Loreau et al. 2001, Hooper et al. 2005, Drever et al. 2006, Bodin and Wimen 2007). Most ecosystem processes are controlled by, or are the result of, biodiversity. However, not all species are equally important in maintaining these processes (Walker 1992, 1995, Diaz et al. 2003) and there is redundancy at multiple levels within most ecosystems (Hooper et al. 2005). Functional species that dominate ecosystem processes are not inevitably the most numerous species (e.g., Hooper and Vitousek 1997, Diaz et al. 2003). Evidence has accumulated implicating the relationship between functional diversity and ecosystem properties, including resilience and the related system attributes of stability and resistance (Diaz and Cabido 2001, Hooper et al. 2005). Under changed conditions, species that had a limited functional role may become functionally dominant, buffering the ecosystem against large changes and enabling resilience (Walker 1995).

Genetic diversity and resilience to environmental change

The basis of all expressions of biological diversity is genotypic variation found in populations. The populations that comprise each level of ecological organization are subject to natural selection and contribute to the adaptive capacity or resilience of tree species and forest ecosystems (Muller-Starck et al. 2005). Diversity at each of these levels must be maintained to facilitate adaptation and foster natural regeneration of forest ecosystems under the climatic changes that have occurred throughout the Quaternary Period (DeHayes et al. 2000), and the anticipated changes from anthropogenic climate warming.

Genetic diversity within a species is important because it is the basis for the natural selection of genotypes within species as they respond or adapt to environmental changes (Pitelka 1988, Burger and Lynch 1995, Schaberg et al. 2008). Genetic-based strategies for reforestation in the presence of rapid climate change must focus on maintaining species diversity and genetic diversity within species (Ledig and Kitzmiller 1992). In the face of rapid environmental change, it is important to understand that the genetic resilience, or adaptive capacity, of forested ecosystems depends largely on the extant, or in situ, genetic diversity within each population of species (Bradshaw 1991). Populations exposed to a

rate of environmental change exceeding the rate at which populations can adapt, or disperse, may become extinct (Lynch and Lande 1993, Burger and Lynch 1995). Species can disperse by seed or vegetative propagules towards a more favourable environment, or they can change gene frequencies to favour genotypes that are better adapted to the changed environment (Burdon and Thrall 2001, Reusch et al. 2005). Species may also adapt through phenotypic plasticity, if their genotype enables a range of responses that are suited to the new conditions (Nussey et al. 2005).

Trees are among the most genetically diverse of all organisms (Hamrick and Godt 1990) and this diversity within natural populations provides the foundation for population stability in variable environments (Gregorius 1996). Concerns have been expressed that predicted climate changes may occur too quickly for species to adapt (Davis and Shaw 2001, Huntley 1991, Jump and Penuelas 2005), but genetically diverse species are capable of rapid evolution (Geber and Dawson 1993). Further, Doi et al. (2009) have noted that high genetic diversity increases the induced capacity of plants to respond to climate change, for example flowering times in plants have already responded to climate change (Franks et al. 2007). Trees are obviously a primary focus for managers in forest ecosystems and so maintaining high levels of genetic diversity maybe a means to help forests respond to environmental change and maintain forest goods and services.

The relationship among biodiversity, productivity and function, and resilience

There is debate over the role that biodiversity plays in ecosystem function and stability owing to the highly complex nature of relationships among species and the synergistic roles of extrinsic factors and intrinsic factors in ecosystems (see e.g., Kinzig et al. 2001, Loreau et al. 2002, for summary discussions). Nevertheless, in the absence of biodiversity there would be no ecosystems and no functioning. Three general reviews from multiple ecosystem types have all found in >65% of the studies negative effects of loss of biodiversity on ecosystem function (Schlapfer and Schmid 1999, Cardinale et al. 2006, Balvanera et al. 2006). Other studies reviewed found no effect until several species were removed. There is evidence that complex forest ecosystems are more productive than simple ones (under the same conditions, see Table 1), and generally that simple forest systems are highly prone to various catastrophes including disease and invasion (e.g., Scherer-Lorenzen et al. 2005).

Two hypotheses predict the relationship between biodiversity and productivity in ecosystems: the niche complementarity hypothesis (Tilman et al. 1996, Tilman and Lehman 2001) and the sampling effect hypothesis (Aarssen 1997, Doak et al. 1998).

Table 1 Summary of published studies in forests that tested the relationship between species richness and some measure of production (biomass, increment, soil C, etc.). Studies testing effects herbicides, thinning, fertilisation, and N-fixing plant facilitation were excluded. Observational (obs) refers to studies where data were gathered from existing forest stands and experimental (expt) refers to directed planting or removal experiments

Author	Forest type	Observational or Experimental	Effect of multiple species on stand production	
			Positive	Neutral
Prokopenv 1976	Boreal	Expt	X	
Ewel et al. 1991	Tropical	Expt	X	
Longpré et al. 1994	Boreal	Obs	X	
Schultze et al. 1996	Temperate	Obs		X
Wardle et al. 1997	Temperate	Expt		X
Parrotta 1999	Tropical	Expt	X	
Montganini 2000	Tropical	Expt	X	
Enquist and Niklaus 2001	Temperate	Obs		X
Casparsen and Pacala 2001	Temperate	Obs	X	
Schroth et al. 2002	Tropical	Expt	X	
Petit and Montagnini 2004	Tropical	Expt	X	
Pretsch et al. 2005	Temperate	Expt	X	
Jones et al. 2005	Temperate	Expt	X	
Vilà et al. 2005	Temperate	Obs	X	
Erskine et al. 2006	Tropical	Expt	X	
Bristow et al. 2006	Tropical	Expt	X	
Finn et al. 2007	Tropical	Expt	X	
Kirby and Potvin 2007	Tropical	Obs		X
Healy et al. 2008	Tropical	Expt	X	
Murphy et al. 2008	Tropical	Expt		X
Piotto 2008	Meta-analysis of 14 plantation studies	Expt	X	

Few studies have tested these hypotheses in highly connected systems with multiple trophic levels and complex production webs, such as forests. While the work from simple ecosystems has limited applicability in forests, it presents theoretical predictions for what species do in ecosystems. The niche complementarity hypothesis predicts that as species are added to a system, the productivity in the system will increase until vacant niches are filled because of effective partitioning of resources. The coexistence of species then is assured through interspecific differentiation in response to competition for resources. If species are able to avoid competition by occupying different niches, then production in the system will increase accordingly (e.g., Tilman and Lehman 2001, Tilman et al. 2002). The sampling (or selection) effect hypothesis, suggests that dominant competitors play the greatest roles in ecosystem functioning and as diversity increases, functioning in the system will be

controlled by these dominant species because of their greater likelihood of being present in a diverse system (e.g., Aarssen 1997, Huston 1997). Various studies suggest support for one or the other of these models (e.g., Hooper and Vitousek 1997, Tilman et al. 2002, Hooper and Dukes 2004). Facilitation also deserves consideration as a possible mechanism, distinct from complementarity, by which increased species richness could enhance production.

Many authors have suggested that it is not diversity *per se* that influences production and resource dynamics but rather it is the number of functional groups. While studies have indicated a link between plant species richness and ecosystem production (Symstad et al. 1998, Wardle et al. 1999, Schwartz et al. 2000, Schmid et al. 2002, Tilman et al. 2002, Hector 2002), species richness and functional richness are not necessarily correlated (Diaz and Cabido 2001, Hooper et al. 2005). Some species

play greater functional roles in systems than others (Walker 1994, Schlapfer and Schmid 1999, Chapin et al. 2000, Diaz and Cabido 2001), but species-specific functional roles may depend on the individual ecosystem (Hooper et al. 2005).

Evidence of a diversity productivity relationship in forests

While most of the work on biodiversity and production is from grasslands, testing these theories of the relationship between diversity, productivity, and resilience in forests is difficult owing to the inability to control either extrinsic or intrinsic variables within complex ecosystems. Furthermore, niche partitioning is well-known in forests (e.g., Leigh et al. 2004, Pretzsch 2005), with many examples such as rooting systems, shade tolerances, xeric and hydric species. Some confounding factors affecting production in forests include successional stage, site differences, and history of management (Vila et al. 2005). Of 21 forest studies reviewed (Table 1) (excluding studies using herbicides, thinning, fertilisation, and N-fixing facilitation), 76% suggested a positive effect of mixed species (i.e., number of species) on ecosystem production. In plantations, the effects of mixing species can be negative owing to competition and so results of such experiments can be directly related to the species mixtures that were selected. On the other hand, facilitation and additive effects on mean annual increment were seen in many studies (Kelty 2006, Piotto 2008), especially in studies where an N-fixing species were included (Forrester et al. 2006, Piotto 2008).

Carbon sequestration, a frequently measured variable among studies, is enhanced by the presence of multiple complex levels of functional groups in forests (studies in Table 1). This notion is further supported by recent studies that complex old-growth forests provide carbon sinks and may continue to do so for centuries, with production, unless disturbed (Baker et al. 2004, Luyssaert et al. 2008, Lewis et al. 2009). Mechanisms of complementarity effects observed in mixed species forests may be nutritional, as a function of improved soil condition (e.g., Ewel et al. 1991, Brantberg et al. 2000, Hattenschwiler 2005), or related to improved partitioning of resources through different rooting patterns and depths (Schmid and Kazda 2001). The evidence broadly supports the concept that diverse forests provide more goods and services than do simple forests, especially monocultures (e.g., Pearce and Moran 1994, Srivastava and Velland 2005, Diaz et al. 2005, Dobson et al. 2006).

Diversity-productivity relationships and forest resilience

Species functional characteristics strongly affect ecosystem properties and the abundance of individual species may not be related to impact in an ecosystem (Diaz and Cabido 2001, Hooper et al. 2005).

Functional diversity in forests relates to production in the ecosystem (Chapin et al. 1997, Diaz and Cabido 2001), and many species in forests appear to be redundant in terms of total production (Pretzsch 2005). Redundancy, also referred to as the insurance hypothesis (Naeem 1998, Yachi and Loreau 1999), appears to be common in natural forests contributing to their resilience, protecting against effects of species loss or responding to environmental change. For example, several tree species have been lost, or substantially reduced in abundance in temperate forest ecosystems with little or no loss of productivity (e.g., Pretzsch 2005), suggesting compensation by other species.

Diversity and stability

Ecosystems respond to environmental change through functional compensation, or the dynamic capacity of systems to maintain production, although levels of output among species may change (e.g., Loreau 2000). This concept is closely linked to that of functional redundancy in diverse ecosystems (Naeem 1998, Yachi and Loreau 1999). Dynamic responses in diverse ecosystems that maintain stability to environmental change over time, may occur at genetic, species, or population levels. There appears to be a low variability among ecosystem properties in response to change in diverse systems compared in to those systems with low diversity where higher variance is observed (Hooper et al. 1995, Ives et al. 1999, Lehman and Tilman 2000, Hughes et al. 2002). Overall, the evidence is consistent with the concept that diversity enhances stability of ecosystem processes (Hooper et al. 2005) and the flow of goods and services.

Forest ecosystems have multiple stable states that depend on the kinds of disturbances that forests undergo (Marks and Bormann 1972, Mayer and Rietkerk 2004, Schroder et al. 2005), but many of these alternative states deliver similar goods and services. Folke et al. (2004) suggested that biodiversity is a slow-changing variable that has consequences for ecosystem state, acting primarily through species with strong functional roles. A major factor impeding the recovery and stability of forest ecosystems is degradation, resulting from unsustainable use and causing the loss of functional species and reduced redundancy, which results in the ecosystem moving to an undesirable state that may have high resilience.

Forests are dynamic mixtures of ecosystems over time and across landscapes. Stability of ecosystem processes in the face of disturbances may be positively related to diversity in these ecosystems (McCann 2000, Ingham et al. 1985, Liiri et al. 2002). Hooper et al. (2005) suggested that the majority of evidence supports the notion that a range of species, which respond in different ways to changes, confers stability to ecosystem processes. However, there is only limited evidence on the relationship between

diversity and stability of production in forests.

A high species richness is also thought to reduce invasiveness of a system. However, invasion of an ecosystem is highly variable among systems and depends considerably on the capacity of the invader, vacant niches available, propagule pressure, available enemies, and other factors, making prediction difficult (Davis 2009). Nevertheless, undisturbed biodiverse tropical systems are less invaded than many other forest types.

Resilience, biodiversity, and forest carbon dynamics

Forest-carbon dynamics are driven by the climatic inputs that govern rates of photosynthesis, respiration, and decay (e.g., Kirschbaum 2004). Rates of photosynthesis scale with increasing water availability, if thermal and radiation regimes are sufficient. Holding wetness constant results in respiration-decomposition rates scaling with temperature; generally, the rate of biochemical processes doubles with every degree Celsius. Differences in the chemical and physical characteristics of substrates also influence growth rates due to locally-scaled variations in sub-surface water availability and soil nutrient status (Law et al. 2002, Chambers et al. 2000). Among biomes, major differences occur in forest carbon dynamics (Keith et al. 2009). Tropical forests have the least dead and soil carbon due to high respiration and turn-over rates associated with increasing temperature, while boreal forests have the converse. Particular forest ecosystems can store significantly more carbon in living and dead biomass as the result of local conditions, and estimates of stocks can be low due to land use history (Keith et al. 2009).

Micro-habitat buffering plays a critical role in all forests, but perhaps reaches its strongest expression in tropical forests (Kennedy 1997, Malhi et al. 2009). Primary tropical forests create a microclimate that virtually eliminates the probability of fire, whereas second-growth forests in the eastern Amazon area were found to burn after 8-10 rainless days (Uhl and Kauffman 1990). The synergistic effects of biodiversity on primary productivity are most evident in primary tropical forests with respect to nutrient cycling. Many tropical forests naturally form on nutrient-poor substrates but can harvest the needed nutrients from rainwater. Furthermore, through retention and recycling, they build up the stock of nutrients to support high levels of plant growth enabled by moist tropical climates. Plants have special adaptations to conserve nutrients and a myriad of other fungal, bacterial, and animal species aid their efficient and rapid recycling (Golley 1983). Hence, biodiversity serves to increase the productivity and resilience of carbon dynamics in tropical forests.

The role of biodiversity in conferring resilience to forest-carbon dynamics varies among climatic

domains, and climate change will alter forest-carbon dynamics by affecting rates of photosynthesis and respiration-decay. However, whether total ecosystem carbon increases or decreases will depend on 1.) the magnitude of increase in temperature, and 2.) the direction and magnitude of change in wetness. While regional trends in temperature can be projected with reasonable reliability, projected regional changes in wetness are highly variable among models for most terrestrial systems (IPCC 2007). However, models suggest significant regional-scaled impacts are likely (Malhi et al. 2009). Large-scale loss of biodiversity will have dramatic negative effects on carbon sequestration capacity by tropical forests (Cramer et al. 2004, Fischlin et al. 2009). Hence, protecting and restoring biodiversity serves to maintain resilience in forests, in time and space, and their ongoing capacity to reliably sequester and store carbon.

Summary

This review, together with those of Loreau et al. (2001), Hooper et al. (2005), and Drever et al. (2006), suggested strong support for the following concepts specific to forest ecosystems:

1. Resilience is an emergent ecosystem property conferred at multiple scales by biodiversity in forest systems. More specifically, resilience over time and space is related to genetic diversity, functional species diversity, and ecosystem diversity of a forest landscape.
2. Natural forests are highly resilient ecosystems, adapted to various perturbations. If disturbance exceeds the capacity of the forest to recover (forest degradation reducing functional components), the system will recover to a different state that may or may not also be highly resilient, but which is unlikely to provide the former level of goods and services.
3. Complex forest ecosystems are generally more productive than simple systems, especially over time and space.
4. There is niche differentiation among forest tree species, leading to complexity and variability within and among forest ecosystems and their processes.
5. Redundancy of functional species is common in complex forest ecosystems and is directly related to ecosystem resilience.
6. Diverse forest systems are more stable than less diverse systems.
7. Although a forest may change states in response to disturbances, the goods and services may not necessarily be highly altered, suggesting resilience even though the species mix and ecology of the system has changed. Such a response is unlikely to occur in a system that has low redundancy.
8. There is a positive relationship between species diversity, landscape diversity, and the capacity of a forest system to be invaded, especially by pests and diseases. This relationship, however, is scale-dependent often resulting in a paradox (i.e.,

negative relationship at small scales).

9. Diverse forests provide more goods and services than less diverse forests, such as carbon sequestration.

10. Not all forest ecosystems are equally resilient to disturbances, including climate change. Effects of climate change will vary in forests depending on biome, tree species composition, disturbance regime, and moisture, temperature, and edaphic condition responses to climate change. Effects of climate change will be particularly negative in tropical rainforests that will experience drying.

All forest types will undergo some change as a result of altered climate conditions; some of these changes are already occurring but widespread change is expected over the next 50-100 years (e.g., Alcamo et al. 2007, Fischlin et al. 2009). Some forests are considerably more vulnerable (less resilient) than others as a result of altered disturbance regimes as predicted under climate change. This is especially the case where previously rarely-seen disturbances will become more common, such as fire in tropical rainforests. In some cases, forests are expected to change states to non-forest or savannah (IPCC 2007). The result generally will be altered forests with a reduced overall capacity to store carbon, except for some boreal forests (Fischlin et al. 2009).

Management recommendations

Forests have a capacity to resist environmental change owing to their multiple species and multiple processes. However, a reduction in biodiversity has implications for system functioning and the amounts of goods and services produced. While it is relatively simple to plant trees and produce a short-term wood crop, the lack of diversity, at all levels, in these systems reduces resilience, degrades the provision of goods and services that the system could provide, and renders the system vulnerable to catastrophic disturbance. Specifically for C sequestration as a mitigation of climate change, long-term stable forests will be of key importance, as opposed to rapidly growing simple forests that have limited resistance and resilience.

Maintaining resilience in forests is important to keep their function in the global carbon cycle by maximizing their potential to sequester carbon and produce other goods and services. Human use of forests will need to change to ensure their conservation. The capacity to conserve, sustainable use and restore forests rests on our understanding and interpretation of pattern and process at several scales, the recognition of thresholds, and the ability to translate knowledge into appropriate adaptive management (Frelich and Reich 1998, Gauthier et al. 2009). The capacity to care for forests in ways that maintains their diversity and resilience is being made even more complex owing to climate change (e.g., Chapin et al. 2007, Kellomaki et al. 2008). Ecological principles to maintain and enhance long-term forest resilience, especially under climate

change, should include (e.g., Thompson et al. 2002, Fischer et al. 2006, Millar et al. 2007, Innes et al. 2009):

1. Maintain stand and landscape structural complexity using natural forests as models and benchmarks.
2. Maintain connectivity across forest landscapes by reducing fragmentation, recovering lost habitats (forest types), and expanding protected areas networks.
3. Maintain functional diversity (and redundancy) and eliminate conversion of diverse natural forests to plantations.
4. Modify management regimes based on modelling of future scenarios.
5. Control invasive species and reduce reliance on non-native tree crop species.
6. Hedge bets by apportioning some areas of assisted regeneration with trees from regional provenances and species from climates that approximate expected conditions in the future.
7. Maintain biodiversity at all scales (stand, landscape, bioregional) and of all elements (gene, species, community).
8. Ensure there are national networks of scientifically designed, comprehensive, adequate and representative protected areas.
9. Pay special attention to the conservation of isolated stands of tree species at the margins of their geographic ranges because of their potential genetic value in maintaining forest resilience (Cwynar and MacDonald 1987, particularly in northern areas).
10. Maintain genetic diversity by not only selecting certain trees for harvesting based on site, growth rate, or form, because selective tree removal can alter gene frequencies, especially among rare alleles (Schaberg et al. 2008).

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Ensuring food production: native biodiversity provides pollination and biological control services

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Abstract

Land use intensification drives extinctions of species and alters the ways in which they interact with one another. This loss of biodiversity may result in reduced rates of ecosystem services such as pollination and biological control, with loss of functional group and response diversity having the greatest effects on function. Biodiversity also acts as insurance in changing conditions, so in addition to reduced mean rates of ecosystem services, stability of these services may also decline. Similarly, alterations to the dynamic structure of networks of interactions among species may affect their resilience to other environmental changes. Conservation of natural forests, as well as ‘softer’ agricultural/silvicultural systems can help to conserve regional biodiversity, which can enhance ecosystem functioning in adjacent managed habitats. Conservation of heterogeneous landscapes, including natural forests, will be necessary to maintain ecosystem services in the face of a suite of interacting global environmental changes.

Keywords: Global environmental change; ecosystem service; insect; land use change; food web.

Introduction: land use change and biodiversity loss

Sixteen years after the Convention of Biological Diversity first came into force, forests and their associated biodiversity continue to decline. Almost 6 million hectares of forest are destroyed each year in the humid tropics alone (Achard et al. 2002). One of the strongest predictors of forest loss is human population density (Wright and Muller-Landau 2006), which continues to grow in most regions, particularly those with high biodiversity (Cincotta et al. 2000).

The primary driver of forest loss is clearance for agricultural land use, with humans now appropriating more than a third of total terrestrial net primary production (Foley et al. 2005, Foley et al. 2007, Haberl et al. 2007). These land use changes take place at multiple scales, with an increasing proportion of available land being sequestered for agriculture, and a concomitant increase in the management intensity of agricultural land (Tscharntke et al. 2006). This produces not only a loss in total forest cover, but also a loss of landscape- and habitat-scale heterogeneity in vegetation structure.

These changes in land use are the greatest driver of biodiversity loss globally (Sala et al. 2000), and it is estimated that current extinctions are occurring at between 100 and 1000 times pre-human rates (Pimm et al. 1995). In addition to the obvious moral tragedy of these losses, elevated extinction rates have engendered concern about the effects of species losses on the functioning of ecosystems and the consequences for human wellbeing (Lawton 1994, Chapin et al. 2000). Clearly the loss of certain species such as crops or medicines (i.e. material goods) could be devastating, and the variety of ecosystem services (Myers 1996, Daily et al. 2000) that natural ecosystems provide to humans are estimated to be worth almost twice the global GNP (Costanza et al. 1997). However, what has been less clear is whether the loss of biodiversity per se is important for maintaining ecosystem services, or whether we should be more concerned about the loss of certain economically or functionally important species.

Here I will summarise the impacts on ecosystem services of lost biodiversity through continued deforestation. Specifically, I will focus on the effects of biodiversity loss on pollination and biological control of pests, two ecosystem services involved in food production (Fig. 1). These services are not trivial, as 76% of our food crops (35% of food volume) depend on animal pollination (Klein et al. 2007), and natural enemies provide an estimated \$4.5 billion worth of pest/disease control each year in the US alone (Losey and Vaughan 2006). In addition to changes in these ecosystem services, I will examine evidence that biodiversity may promote resilience in services under changing environmental conditions. Finally, I will discuss strategies for conserving functional diversity in order to maximise the provision of ecosystem services.

Biodiversity and ecosystem services

The last decade and a half has seen an exponential increase in the number of studies examining the effects of biodiversity on rates of ecosystem functions or processes (Fig. 2). Many of the early studies were conducted in experimental grassland plots, and generally found a positive effect of species diversity on plant productivity (Hooper et al. 2005). These effects can sometimes be due to the fact that more diverse assemblages are more likely to contain, by

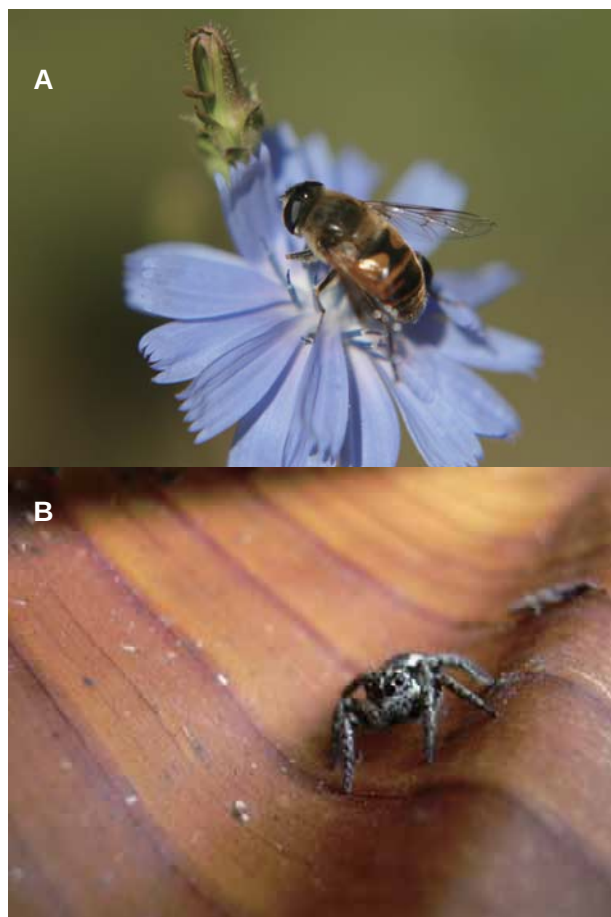


Fig. 1 Ecosystem service providers. A) Pollinators such as this hoverfly (in an apple orchard, Region del Maule, Chile) can be critical for the reproduction of both crop and wild plants. B) Predators such as spiders (here in a coffee plantation, Manabi, Ecuador) can be important regulators of insect pest populations.

chance alone, a particularly productive species (Huston 1997, Cardinale et al. 2006). This lottery or 'selection' effect would suggest that, provided we know which species are important, the loss of other species would have little impact on ecosystem functioning. However, not only would this assumption ignore the likelihood that species can become more or less important at different times or under different conditions (see 'Biodiversity as insurance in fluctuating environments' section below), but there is also evidence that species diversity per se can affect primary productivity through niche complementarity (Hooper et al. 2005, Cardinale et al. 2007). Under this mechanism, different species utilise slightly different resources, or obtain them in slightly different ways. This reduces competition within species, allows fuller exploitation of the available resources/niches, and maximises rates of an ecosystem process (Hooper et al. 2005, Cardinale et al. 2007).

In addition to this work on plant biomass production, recent evidence suggests that high diversity of animals that provide ecosystem services, such as biological control and pollination, can

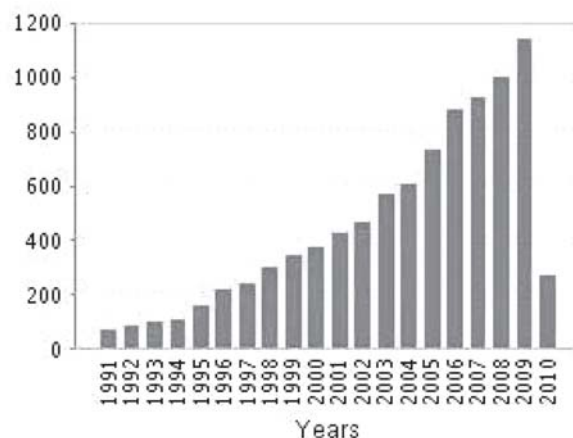


Fig. 2 Increasing number of studies testing effects of biodiversity on ecosystem functioning per year. Data compiled from Web of Science database on 14 April 2010 using search terms (biodiversity or "species richness" or "species diversity") and ("ecosystem function*" or "ecosystem service" or "ecosystem process" or "productivity" or "primary production" or biomass or "parasitism rate" or "predation rate" or pollination).

enhance food production. For example, Cardinale et al. (2003) showed that diverse insect predator communities were twice as effective as single species at reducing populations of aphids on alfalfa, and this improvement in biological control even doubled alfalfa yield. Similarly, Snyder et al. (2006) found that diverse control agents were more effective at controlling two species of aphids on collards, and that reduced aphid densities resulted in increased plant growth. Diversity of animal pollinators can also improve fruit set and crop yields. For example, high diversity of pollinators has been shown to increase pollination success and yield of a variety of food crops, including coffee (Klein et al. 2003), tomato (Greenleaf and Kremen 2006), and watermelon (Kremen et al. 2002).

The benefits of biodiversity are greatest when species differ in their functional characteristics, thereby providing diversity of functional groups as well as species (Elmqvist et al. 2003, Luck et al. 2003). For example, pollinator assemblages where several species differ significantly in their morphology and/or pollinating behaviour can be most effective at pollinating crops, particularly when the flowers are variable in their location or time of opening (Hoehn et al. 2008). A recent study of pumpkin grown in Indonesian homegardens found that pollinator species differed in their functional traits, such as preferred pollinating height, time of day at which they were most active, body size, and the way in which they carry pollen. Therefore, diverse pollinator assemblages contained a greater diversity of these traits, and were consequently able to pollinate all the pumpkin flowers more successfully than assemblages with fewer species (Hoehn et al. 2008). In fact, when

Hoehn et al. (2008) used multivariate methods to group pollinator species into functional groups based on their physical and behavioural differences, the diversity of functional groups was a stronger predictor of pollination success (the number of seeds per pumpkin) and yield (the size of the fruit), than was pollinator species diversity.

Such examples provide evidence that biodiversity loss can have important consequences for real-world food production. However, for niche complementarity to occur, there must be a variety of niches available to be partitioned among functional groups (e.g., the different height and timing of flowers in the study of Hoehn et al. 2008). Therefore, we may hypothesise that the effects of biodiversity on ecosystem process rates will be greatest when habitats or resources are heterogeneous, i.e. when there are a variety of different resource niches available. This hypothesis was tested recently for three different ecosystem processes in three different systems: biomass production in German grassland communities, parasitism rates by wasps in a range of habitats in Ecuador, and coffee pollination in Sulawesi, Indonesia (Tylianakis et al. 2008b). The authors found that, in all three cases, the positive effect of biodiversity (of grasses, parasitoids, and pollinators) on process rates (biomass production, parasitism, and pollination respectively) increased with increasing heterogeneity of the limiting resource (soil nutrients, host larvae, and coffee flowers; Tylianakis et al. 2008b). This suggests that the benefit of biodiversity for ecosystem services such as pollination and biological control will be greatest in heterogeneous natural and seminatural ecosystems, such as those found in agroforests and forests.

In addition to the direct benefits of animal (pollinator and natural enemy) diversity for pollination and biological control, plant biodiversity within or adjacent to production systems can provide benefits for food production. First, diverse plants provide a variety of floral resources to sustain diverse pollinator communities, and an array of herbivorous insects to provide prey for natural enemies. Thus, it is not surprising that diversity of herbaceous plants has been shown to correlate positively with diversity of bees and wasps (Tylianakis et al. 2006a), or that pollinator diversity can help to maintain diverse plant communities (Fontaine et al. 2006). These insect species can be abundant in forests, but also move out into adjacent crops, providing high rates of pollination (Ricketts et al. 2004, Blanche and Cunningham 2006, Blanche et al. 2006; but see Chacoff et al. 2008) and biological control (Landis et al. 2000) close to forest habitats. In addition to the benefits of plant diversity for maintaining abundant animal service providers, plants may also increase the per-capita efficacy of these animals. For example, many parasitoids of pest insects feed on the pest during their larval stage, but require floral nectar during their adult phase. Providing non-crop floral resources adjacent to crops

can enhance the longevity and fecundity of parasitoids, thereby enhancing biological control (Tylianakis et al. 2004). This phenomenon is well-studied in arable crops, though its potential for the enhancement of biological control in forests and agroforests has received less attention.

Biodiversity as insurance in fluctuating environments

The above examples highlight the importance of biodiversity for maintaining high (average) rates of functions/services such as biological control or pollination. However, as important as having high rates of food production, is the necessity of stability of food resources, both through time and in the face of environmental changes. The effects of biodiversity on stability have received considerable research attention (May 1973, McNaughton 1978, Givnish 1994, Hanski 1997, McGrady-Steed et al. 1997, Hughes and Roughgarden 2000, Worm and Duffy 2003, Hooper et al. 2005, Ives and Carpenter 2007), with measures/definitions of stability varying widely across studies (Pimm 1984, Grimm and Wissel 1997). In terms of pollination and biological control services, greatest attention has been paid to reducing temporal variability. By occupying distinct temporal niches, pollinator or natural enemy species can cause, through statistical averaging, a reduction in temporal variance of pollination or pest control (Yachi and Loreau 1999). For example, Kremen et al. (2002) found that the importance of different wild pollinator species changed from year to year, meaning that sites with higher pollinator diversity experienced sustained pollination through time. Similar effects have been shown for attack rates by insect parasitoids, where temporal variability is reduced by parasitoid diversity (Tylianakis et al. 2006b). Biodiversity can also theoretically increase the resilience of ecosystem services (Petersen et al. 1998, Elmqvist et al. 2003) by buffering against environmental change (McNaughton 1978). The strength of this buffering or 'insurance' effect depends on the degree of asynchronicity in the responses of individual species to environmental fluctuation, and on the specific nature of their responses (i.e. response diversity; Yachi and Loreau 1999, Elmqvist et al. 2003).

It is also important to recognise that the world is experiencing a suite of environmental changes simultaneously (Sala et al. 2000), and that the effects of these changes on ecosystems may not be independent of one another (Folke et al. 2004, Tylianakis et al. 2008a). For example, land use intensification may allow generalist invasive species to become dominant and further affect native biodiversity (Didham et al. 2007). Ultimately, any strategies for conserving biodiversity will need to consider the interrelated nature of the drivers of species loss (Folke et al. 2004), and recognise that mitigation of the effects of one driver may require

actions to reduce another (Didham et al. 2007).

Analogous to these temporal insurance effects is the spatial insurance effect of biodiversity in patchy environments (Loreau et al. 2003). In mosaic landscapes, different species occupy different habitat types. This turnover of species among habitats is called beta diversity, and can contribute significantly to the overall (gamma) diversity of a region (Tylianakis et al. 2005). In addition to providing high overall diversity, beta diversity may be important for maintaining spatial insurance in pollination or biological control (Loreau et al. 2003). Similarly, even when a given species occupies multiple habitats, it can move between patches, becoming an important service provider in some patches or at certain times (e.g., promoting recovery following disturbance; Lundberg and Moberg 2003), even though it may be less abundant or important in others (Srivastava and Vellend 2005). Thus, spatial and temporal occurrence and turnover in biodiversity can provide resilience in ecosystem services (Bengtsson et al. 2003, Tschamntke et al. 2007), but species turnover in time and space may be reduced by land use intensification at the landscape scale (Tylianakis et al. 2005).

Interactions between species

Many ecosystem services (including pollination and biological control) involve interactions among two or more species. Interactions between species are determined by the relative abundance of different participant species, their behaviour, phenology, etc, and this vulnerability may mean that we observe changes in species interactions before the species involved actually go extinct (Janzen 1974, Tylianakis et al. 2008a). A recent review of almost 700 studies measuring responses of over 1000 pairwise species interactions to different drivers of global environmental change found that pollination interactions tend to decline in strength or frequency with land use change, particularly habitat fragmentation (Tylianakis et al. 2008a). In contrast, effects on insect predator-prey and parasitoid-host interactions were much more variable, making future changes in the success of biological control difficult to predict. Even more variable was the change in interactions with different drivers of environmental change. Pollination tended to be negatively affected by all drivers tested, but responses of insect natural enemy interactions varied considerably across drivers (Tylianakis et al. 2008a). As mentioned above, the effects of these drivers are not independent, and the modification of habitats can alter interactions between invasive species and their native competitors, further facilitating invasion (Didham et al. 2007). For example, intensification of cacao agroforests can promote invasion by exotic ants, which then reduce the diversity of native forest ants disproportionately compared with their effects on habitat generalist species (Bos et al. 2008). Although this study did not

test functional effects of ant invasions, loss of forest ant biodiversity could potentially result in a decline in the functions/services they provide.

Despite the importance of changes to biodiversity or mutualistic and antagonistic interactions, pairwise interactions between species do not occur in isolation. Rather, they are components of a larger network of feeding and/or mutualistic interactions, whose structure can be critical for ecosystem stability (May 1973, Paine 1988, Dunne et al. 2002, Kondoh 2003, de Ruiter et al. 2005, Bascompte et al. 2006, McCann 2007), for example, by determining the effects of species extinctions on community-wide pollination success (Bascompte et al. 2003). Various attributes of the structure of these networks can therefore have important implications for conservation (Tylianakis et al. 2010) but the impacts of this structure on stability cannot be predicted from the pairwise interactions alone.

Thus, in addition to local or global extinctions of species, there may be less obvious, insidious effects of agricultural change on the interaction structure of ecosystems. Although methods for quantifying these interaction networks have been around for more than a decade, it was unclear until recently what effect, if any, land use changes have on food web structure. A study of 48 quantitative networks of feeding interactions ('food webs') involving bees, wasps and their natural enemies in Ecuador, showed for the first time that the species interactions comprising the "web of life" are sensitive to changes in land use (Tylianakis et al. 2007). Conversion of forests to intensive agriculture led to a sizeable shift in food web structure, and subsequent dominance of the webs by one or two interactions resulted in bees and wasps suffering heavy attack rates from their natural enemies (particularly an introduced parasitic wasp). This increase in parasitism of pollinators and insect predators could have significant effects on the ecosystem services that these species provide.

A number of subsequent studies have examined the responses of networks of parasitoid-host or pollinator-plant interactions to land use change and habitat fragmentation (Albrecht et al. 2007), species invasions (Lopezaraiza-Mikel et al. 2007, Aizen et al. 2008), and climate change (Memmott et al. 2007). Unfortunately, pollinator-plant networks are usually examined independently from parasitoid-host networks (but see Henson et al. 2009 for an impressive exception), when the interplay between these two functions may in fact determine crop productivity. Moreover, in addition to altering the overall structure of the network, anthropogenic disturbances such as land use changes may homogenize the dynamic structure (i.e. spatial and temporal variability) of networks at the regional scale (Laliberté and Tylianakis 2010). By splitting the 48 parasitoid-host food webs of Tylianakis et al. (2007, see above) into monthly time steps, Laliberté & Tylianakis (2010) found that the structure of

interactions within the webs was more similar across sites and through time in the (most intensive) rice and pasture habitats. Conversely, forested sites (including managed and abandoned coffee agroforests) had network structures that were highly variable through time and space. This reduction in spatial and temporal interaction turnover with land use intensification is analogous to the reduction of species turnover (beta diversity, see above), and could reduce the insurance value (and resilience) of networks in heterogeneous landscapes or under changing conditions.

Strategies for conserving functional biodiversity

The above sections have provided evidence for the widespread loss of biodiversity, ecosystem services, and change in interaction structure of communities following deforestation and land use intensification. The simplest strategy for stemming this tide of extinctions and loss of ecosystem services would be widespread reforestation and reduction in the extent and intensity of agricultural management. Obviously, the pressures of a growing human population will prevent this from occurring everywhere, so we must be pragmatic in our conservation approaches. Usually only regions with high per capita GDP can afford restoration programmes, producing a strong correlation between a nation's wealth and whether its annual net change in forest cover is positive or negative (Ewers 2006). Therefore, developing countries could be less likely to receive any of the ecosystem service benefits (discussed above) that forest preservation may bring, even though they may be more dependent on these services due to the costs of attempting to replace them with pesticides or machinery.

The first priority must obviously be maintenance of as much of the remaining natural forest cover as possible. In addition to the conservation of protected 'set-aside' areas (which cannot be substituted), softer agricultural or silvicultural practices can provide significant benefits for biodiversity and the ecosystem services associated with food production. For example, appropriately managed organic agriculture can sustain higher diversity than conventional systems for a variety of taxa (Bengtsson et al. 2005). It is often thought that this comes at a cost of reduced yields, but Badgley et al. (2007) compared yields of organic vs. conventional systems in different food categories for 293 datasets. For most categories, the ratio was slightly < 1 for developed countries and > 1 for developing countries (Badgley et al. 2007), indicating that organic crops may even deliver higher yields, in addition to higher per unit revenue. Moreover, certain agricultural systems may be inherently better reservoirs for biodiversity. For example, agroforests (Fig. 3) are often characterised by high species diversity (Perfecto et al. 1996, Moguel and Toledo 1999), including large numbers of specialist species



Fig. 3 Agroforest systems. A) Traditional shade-grown coffee (Manabi, Ecuador) can be a valuable refuge for biodiversity, but B) intensification by removal of shade trees (Boquete, Panama) may reduce these benefits.

(Tylianakis et al. 2005), and food web structures that do not differ significantly from those of natural forests (Tylianakis et al. 2007, Laliberté and Tylianakis 2010). In fact, coffee agroforests have even been found on occasion to contain higher species diversity than native forest (Teodoro et al. submitted). However, the benefits of systems such as agroforestry for biodiversity will depend on the way in which they are managed. Clearing of flowering herbs from the ground of coffee agroforests could reduce bee and wasp diversity (Tylianakis et al. 2006a), and reducing the diversity and density of shade tree species in cacao agroforests can cause a reduction in bee diversity (Tscharntke et al. 2008) and increase the spread of, and harm caused by, invasive species (Bos et al. 2008). Thus, even systems that are inherently 'biodiversity friendly' may cease to generate benefits when managed intensively (such as during the transition from shade- to sun-grown coffee, Fig. 3).

Clearly, not all agricultural systems can harbour high levels of functional biodiversity. Yet, appropriately managed agricultural landscapes can still benefit ecosystem services (Tscharntke et al.

2005), and landscape diversity effects may even overwhelm local (farm) scale management (Schmidt et al. 2005). For example, the species associated with natural forests or even non-intensive agroforests can move into adjacent more intensive systems, providing ecosystem services (Landis et al. 2000, Ricketts et al. 2004). Therefore, the maintenance of these habitats within a mosaic landscape may provide benefits beyond the area that they require. However, it is worth noting that the conservation value of these systems (particularly native forests) may suffer through time as pest, weed or predator species that are abundant within the crop spill over into more natural systems (Rand et al. 2006). Finally, even landscapes that do not contain natural habitats can still be managed to maximise biodiversity and resilience. By maintaining a variety of crops, and hence a heterogeneous mosaic landscape, beta diversity among crop patches may contribute to high regional diversity of beneficial species (Tylianakis et al. 2005, Tylianakis et al. 2006a, Tscharntke et al. 2007), and provide a ready source of ecosystem service providers to recolonise after disturbance (Lundberg and Moberg 2003), thereby promoting resilience. In fact, conservation programmes such as the European 'agri-environment' schemes may even be most effective in partially modified landscapes (Tscharntke et al. 2005).

Conclusions

The societal and economic impacts of biodiversity loss (Costanza et al. 1997, Chapin et al. 2000, Daily et al. 2000) are only just beginning to be felt. Here I have focused on two ecosystem services (pollination and biological pest control) that are critical for food production, yet there are many more services that will be affected (Myers 1996, Daily et al. 2000). Not only are the average levels of ecosystem services affected by biodiversity loss, but also their stability through time and in response to environmental changes are likely to suffer (e.g., Elmqvist et al. 2003). Land use intensification is one of several drivers of global environmental change, the effects of which are likely to be interactive in many cases, potentially causing self-reinforcing feedbacks between biodiversity loss and one or more other drivers (Chapin et al. 2000, Didham et al. 2007). Conservation of remaining natural forest habitats will be necessary for slowing the global loss of biodiversity. In addition, softer, 'wildlife-friendly' (Green et al. 2005) forms of agriculture, including agroforestry, can be used as an additional refuge for biodiversity, provided that they are managed appropriately. Finally, entire landscapes must be managed as heterogeneous mosaics to maximise spatial and temporal insurance in ecosystem services (Loreau et al. 2003, Lundberg and Moberg 2003, Tscharntke et al. 2005).

Invariably, expansive monocultures that create a homogeneous environment at the farm and landscape scale will be inimical to the maintenance of ecosystem

services and their stability in the face of foreseeable changes to global environmental conditions.

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Valuation and of ecosystem services: An assessment of conceptual underpinnings

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Abstract

The paper highlights the importance of ecosystem services and biodiversity in enhancing societal welfare. The paper explores the rationale for economic valuation of ecosystem services in the wider context of conservation policies. Subsequently, the paper assesses the theoretical construct of valuation and its implicit challenges. Building upon the latest literature in the field, the paper draws some important lessons learned for robust and credible valuation of ecosystem services.

Key Words: Ecosystem services, valuation, valuation methods

1. Introduction

Humans derive a large number of benefits from different ecosystems. Ecosystems and biodiversity provide a wide range of services through its bio-geo-chemical processes that are critical for sustenance of humans. Ecosystem which is a dynamic complex of plant, animal and microorganism communities and other nonliving environment interacting as a functional unit, provides services which sustain, strengthen and enrich various constituents of human well being. The flow of tangible and intangible benefits e.g. food grain, water, fuel-wood, fodder, nutrient cycling, waste assimilation and climate regulation etc are derived from a well functioning and robust / resilient ecosystem. Some of the direct and tangible benefits are well understood by planners and decision makers, some of the intangible benefits are poorly understood and are left out in the decision making process. The unique feature of most of the services emanating from ecosystems is that they although acknowledged by people, remain unaccounted, unpriced and therefore remains outside the domain of the market. In conventional parlance, such problems are treated as externalities where market fails, and decision makers try to correct the market failure by creating market like situation. Subsequently they obtain the value of services through various valuation techniques based on stated preference of the people. In case of regulating services of ecosystem like climate regulation, waste treatment capacity, nutrient management and various watershed functions, classic situations of market failure appears. The missing market for the ecosystem services adds to the problem because most of the vulnerable section of society primarily in the developing countries and in

Africa and Asia depends upon those services directly or indirectly for their livelihood. Therefore, any decision proves to be inefficient and infeasible from social perspective causing problem for sustainability and human well-being. In recent years, there have been added focuses on creating a situation where market can be created and a desired outcome can be achieved in terms of implications of different decisions culminating through the impact on ecosystems and in turn human well being. In designing the response option, market based mechanisms especially payment / compensation for ecosystem services seems to be emerging as one of the sought after choices by the decision makers (Kumar and Muradian, 2009). Precise and credible economic valuation of ecosystem services strengthens the functioning of its market. This also provides a transparent mechanism not only for acceptable payment mechanism in place but sustainable management of ecosystem as well. Estimation of economic value of ecosystem services has led to the creation of situation where payment for different type of ecosystem services seems to be a consensus response for management of ecosystems.

2. Rationale and Concept of Valuation

Economic valuation of ecosystem services helps in identifying and resolving the trade-offs among different stakeholders engaged in management of ecosystems. Ecosystem management plans often result in net gains for some sections of society and net losses for others. For example, forest conservation through declaring a forest patch as protected forest might increase carbon sequestration (a global benefit) but as a result, local populations might be deprived of access to the forest and be unable to access services, like timber and non timber forest products, as a result. Similarly, for floodplain wetland ecosystems, their conversion might increase the availability of land for agriculture and industrial uses (depending upon the location of the wetland) but services like bioremediation, water storage and biodiversity may be lost, impacting the local poor who might depend upon them. Valuation of ecosystems services is a tool that can help to ensure that the decision-making process incorporates considerations of equity and sustainability.

Economic valuation of ecosystem services helps to link conservation strategy with mainstream policies at national and regional levels. For any ecosystem

service, its social value must be equated with the discounted net present value of the flow of that service (Hanley and Barbier, 2009). Decision-makers can then see how the marginal benefit, for example of conservation of urban or coastal wetlands, equates with the marginal costs of conservation. Estimating the economic value of services like timber and fish, known as provisioning services, is relatively easy because they enter the domain of the market. However, this is not generally the case for regulating services, which can be defined as the benefits people obtain from the regulation of ecosystem processes, including, for example, the regulation of climate, water and some human diseases (MEA, 2005, pp 897).

One area of confusion in the valuation of regulating services has been the decision on what should be valued. Biogeochemical processes and subsequent functions of an ecosystem create services, but not all of these services are appropriated by society. Only those benefits that people obtain from ecosystems should be considered as services (MA 2003, 2005). Thus, valuation should target final rather than intermediate services (Fisher and Turner, 2010). Most of the regulating services are public goods and intermediate in character, but some of the services, like groundwater flow maintained by forests, could be used by lowland people for drinking (consumption) or industrial use (production).

Economic valuation can be defined as an attempt to assign quantitative values to the goods and services provided by environmental resources. The economic value of any good or service is generally measured in term of what we are willing to pay for the commodity less what it costs to supply it. Sometimes, it is construed that economist's approach is to put a dollar value to every natural resources which in any case the society has been considering worthy enough. That is not the case in reality. Economist make an attempt to assess how much society would have to forego for saving a little more of the ecosystems. Obviously, economists are talking about the marginal values of the ecosystem services. There is popular method of 'Total Economic Value' (TEV) of ecosystems which is essential on marginality yardstick but for several functions, an ecosystem is capable of providing to the society.

Valuation is only one element in the effort to improve the management of ecosystems and their services. Economic valuation may help inform management decisions, but only if decision-makers are aware of the overall objectives and limitations of valuation. The main objective of valuation of ecosystem services is to generally indicate the overall economic efficiency of the various competing uses of a particular ecosystem. That is, the underlying assumption is ecosystem resources should be allocated to those uses that yield an overall net gain to society, as measured through valuation in term of the economic benefit of each use adjusted by its costs.

3. Ecosystem Services and Valuation Techniques

All ecosystems whether it is forest, wetlands, mountain, coastal, marine or desert are like any capital stock. They through their ecological production function analogous to engineering production function in production economics, provide ecosystem services. Forest providing the ground water augmentation and carbon sequestration, wetland providing the bioremediation and water storage function, mountain yielding hydrological services etc is some of the examples of ecological services which are beneficial to the society through enabling of production and consumption processes. Various market and non market based valuation methods capture the ecological services in monetary terms enabling them to be incorporated in the box –'values'. There are direct benefits of ecosystems known as intrinsic values or bequest values they directly enter into the 'values' box. Please see the diagram 1.

Formation of values will be influenced by how robust and accurate the valuation methodologies are in capturing the services from the ecological production functions. Values in turn, would determine the human choices. For example, decision making criteria like costs benefits or multi criteria method would depend upon the values arrived through the valuation methods. These decision making criteria would influence the choice which subsequently would impact the condition and trend of the ecosystem in consideration. Here it is very clear that value determine the human choice and the human choice would determine the fate of ecosystem and their services. Therefore, a chain gets established. Valuation of ecosystem plays a pivotal role in the designing the appropriate response option.

Many ecological services and ecosystems are complex and multifunctional, and it is not obvious how the myriad goods and services provided by them worthwhile to deplete or degrade environmental

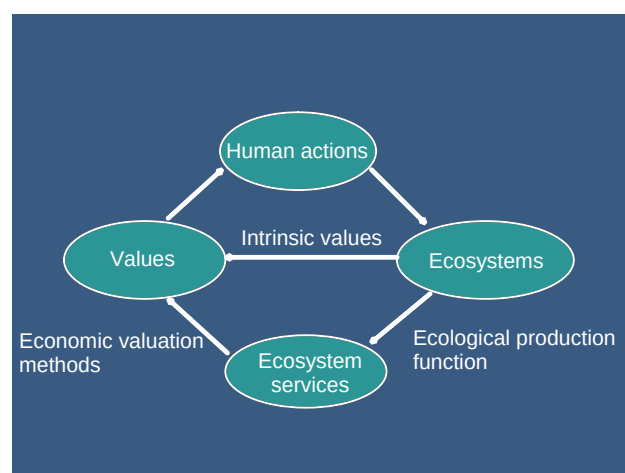


Diagram 1: Ecosystems, values and valuation

resources; in others, it may be necessary to ‘hold on’ to these resources. Economic valuation provides us with a tool to assist with the difficult decisions involved. Loss of environmental resources is an economic problem because important values are lost, some perhaps irreversibly, when these resources are degraded or destroyed. Each choice or option—to leave a resource in its natural state, to allow it to degrade or convert into another use—has implications in term of values gained and lost. This requires that all the values that are gained and lost under each resource use option be carefully considered.

Economic valuation of ecosystem services is instrumental, anthropocentric, individual-based, subjective, context-dependent, marginal and state-dependent (Goulder and Kennedy, 1997; Baumgartner et al 2006, Kumar & Kumar, 2008; Barbier et al, 2009). The value of ecosystem services considers the impact of small change in the state of the world and not the state of the world itself. For example, the marginal value of one unit of ecosystem service does not depend on the total value. The economic value of any asset, including a natural asset like an ecosystem, is only perceived and revealed where the flow of services proves to be beneficial to people. People would be willing to pay for services when they have to incur costs to get these services from alternate sources. The value of ecosystem services is essentially a marginal concept arising out of scarcity and depends on the ecosystem condition and the social-cultural context in which people make choices. Thus, those undertaking valuation should focus on ecosystems that are socially important, evaluate ecological responses in economic value-relevant terms and consider the possible use of a broad range of valuation methodologies to estimate values (EPA, 2009).

There are various market and non-market-based approaches for valuation of ecosystem services and these have been discussed elsewhere (e.g. Freeman, 2003; Heal, 2005; Hanley and Barbier, 2009; Naeem et al., 2009; Kumar, 2010) For valuation of regulating services, market-based, stated preference and production function-based methods have been proposed. The choice of method is influenced by the availability of data, the unit of benefits required, the types of beneficiaries and the expertise of those applying the method. Whichever method selected, it is essential that its application is interdisciplinary (Daily, 1997; Bjorklund et al 1999; MA, 2003 and 2005; Heal 2005; Balmford, 2002; Chichilnisky and Heal, 1998, Ricketts, 2004, Freeman, 2003, Maler et al 2008, 2009; Hanley and Barbier, 2009; Naeem et al, 2009; Barbier, 2006 and 2007; Johnston et al, 2006).

4 Basic Assumption behind Valuation

In valuation of ecosystem services, the utility that an individual derives from a given ecosystem service is assumed to be dependent upon that individual's

preferences. The utilitarian approach, therefore, bases its notion of value on attempts to measure the specific utility that individual or the society derive from a given service, and then aggregates across all individuals, weighting them all equally.

Utility cannot be measured directly. In order to provide a common metric in which to express the benefits of diverse services provided by ecosystems, the utilitarian approach usually attempts to measure all services in monetary terms. This is purely a matter of convenience, in that it uses units that are widely recognized, saves the effort of having to convert values already expressed in monetary terms into some other unit, and facilitates comparison with other activities that also contribute to societal well being. It explicitly does not mean that only services that generate monetary benefits are taken into consideration in the valuation process. On the contrary, practically all work on valuation of environmental and natural resources has been, in essence, to find ways to measure benefits which do not enter markets and so have no directly observable monetary benefits.

The issue of valuation is inseparable from the choices and decisions we have to make about ecological systems. There are some views advocating that valuation of ecosystems is either impossible or unwise, that we can't place a value on such “intangibles” or long-term ecological benefits. Valuation of ecosystems services are not out of luxurious and leisurely activities, it is under dire need for coming out with efficient choice bettering off the state of ecosystems and people dependent upon it. Invariably, the valuation of ecological system is done when decision makers are confronted with the situation of trade off, and competing resources. Valuation has a limited objective to achieve and it will be pity if the exercise of valuation is thrown to the dilemma of ethics, intergenerational/intra-generational equity and other value loaded issues of philosophy and ethics.

5. Methodological Construct

The choice of valuation methods for estimating the value of ecosystem services depend upon data availability, unit of benefits, types of beneficiaries and expertise existing in using that particular methodology. What is important and the key to the success of these methods that, the whole application issue is essentially interdisciplinary in nature. Economist must learn and interact with ecologist if they want to apply their tools meaningfully. Economist would need to learn about the ecological production functions by interacting with limnologist (wetland), plant taxonomist (biodiversity), hydrologist (water recharge) and many other similar professionals found typically outside the circle of economists.

For each typical ecological models starting from individual level to ecosystem levels and corresponding ecological outputs translated into

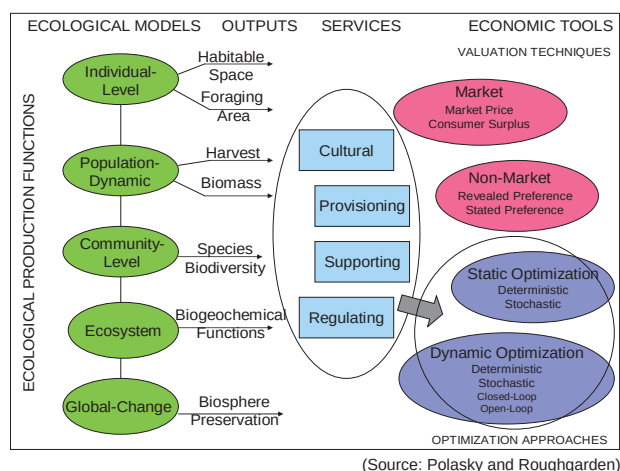


Diagram 2: Ecological models and valuation tools

ecosystem services (provisioning, regulating, cultural and supporting), there would be economic tools (valuation techniques). Following diagram 2 provides the glimpse of the whole scheme

Here it should be clear that as we move from provisioning to regulating to cultural services, valuation methods move from market to non market method. Benefits become public and to capture them into monetary term becomes increasingly difficult.

In the whole discussion of valuation of ecosystem services useful for human well-being and societal welfare, the assumptions of rational economic agents, well functioning markets, consistent preference, straighten choice, learning about the services of ecosystems, and speculations about future seem to be critical. However these assumptions are far from resolved and need serious attention if the value is to be comprehensive and acceptable to all types of professionals. In the past, assumptions of economic theory have maintained distance from behavioral science especially psychology. Economists whether dealing with the issues of valuation or forecasting seem to be functioning independent of the psychological dimension, which is quintessential to the entire exercise of economic analysis of ecosystems. The following section highlights the lacunae in approaches of conventional environmental economics. It offers a fresh perspective on some of the fundamental assumptions of economic sciences that are applied in the valuation of ecosystem services.

6. Challenges in Valuation of Ecosystem Services

Several issues pertinent to valuation of ecosystem services and application to decision making have emerged especially with a better understanding of the mechanisms of ecosystem functioning. The relevance of the state of ecosystem functioning has not been given adequate emphasis in derivation of ecosystem values, thereby rendering the values of little worth, when one is examining issues, especially related to

sustainability.

In order to provide a true and meaningful scarcity indicator of ecosystem values and functions, economic valuation should account for the state of ecosystem. Though, ecosystems can recuperate from some shocks and disturbances through an inherent property of resilience, there are several circumstances under which the ecosystem shifts to an entirely new state of equilibrium. Standard economic theory based concepts deriving ecosystem values based on marginal analytic methods are limited to situations when ecosystems are relatively intact and functioning in normal bounds far away from any bifurcation. This is of particular significance to developing countries, wherein significant tradeoffs exist between conservation and economic development, and decisions often favor the latter. The second issue primarily deals with aggregation of individual values to arrive at larger values, viz. "societal values". Ecosystem goods and services, by definition, are public in nature, meaning thereby that several benefits accrue to society as a whole, apart from the benefits provided to individuals (Daily, 1997; Wilson and Howarth, 2002). The theoretical fundamentals of development of economic valuation methodology rest on the axiomatic approaches of individual preferences and individual utility maximization, which does not justify the public good characteristic of ecosystem services. Valuation methodologies, viz contingent valuation utilize individual preferences as basis of deriving values subsequently used for resource allocation of goods largely public by character. A considerable body of recent literature therefore favors adoption of a discourse-based valuation (Wilson and Howarth, 2002). The primary focus of these approaches is to utilize a discourse based valuation approach to come up with a consensus societal value of scarcity indicator, derived through a participatory process, to be used for allocation of ecological services, largely falling into the public domain.

Recent research effort relating to the valuation of biodiversity and ecosystem services justify the rationale and need for valuation especially for designing conservation strategy and plan (IUCN, 2004; MA, 2005; Barbier, 2009, TEEB 2009, Kumar and Wood 2010, Bateman et al 2010). However, a number of major challenges remain. First, the whole approach to valuing ecosystem services assumes that the linkages between the structure and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values can be identified and quantified and, importantly, meaningfully described to the public (Heal et al., 2005; Barbier, 2007). This challenge is further compounded by the public's low level of understanding of ecosystem services. Third, there is often incomplete knowledge of how ecosystems work and in particular there are issues relating to how valuation can deal with uncertainty, irreversibility and non-linearity in ecosystem functions. Third, there are

a number of issues relating to the aggregation of values, including how to avoid double counting. Fourth, it is recognised that undertaking original economic valuation studies is often impractical, costly and time consuming. An alternative approach is benefits transfer, however, the extent to which policy makers can make effective use of benefits transfer for ecosystem services is unclear.

The first challenge may be addressed through the adoption of an ecosystems approach (MA, 2005)) which helps to explicitly link the ecosystem functions associated with biodiversity to ecosystem services and to values. The second challenge requires the adoption of an inter-disciplinary approach to research where economists and ecologists work together as a cohesive team. Furthermore, the valuation framework needs to be designed so as to explicitly account for any uncertainties in ecological knowledge. Issues relating to aggregation can, to some extent, be accounted for during the design and administration of the valuation protocol; however, it will also be important to test for the impact of aggregation through more qualitative analysis of responses. This also requires an inter-disciplinary collaboration between economists, other social scientists and ecologists.

Application of conventional fundamentals of economic valuation becomes further constrained when sustainability and social equity are also included as goals along with economic efficiency for ecosystem management. While the methodologies for deriving values with economic efficiency as goals is comparatively well developed, integrating equity and sustainability requires a better understanding of functional relationships between various parameters and phenomena responsible for provisioning of the services in the first place and the social processes governing the mechanism of value formation

(discourse based valuation being one such approach).

7. Synthesis and Conclusions

All ecosystems can be considered as capital stock. Through their ecological production function, analogous to engineering production function in production economics, they provide ecosystem services. Forests providing ground water augmentation and carbon sequestration, mountains yielding hydrological services and wetlands providing bioremediation and water storage, are some examples of ecosystem services that are beneficial to society through enabling of production and consumption processes. Various market and non-market based valuation methods capture these ecosystem services in monetary terms, enabling them to be incorporated in the box –‘values’. There are direct benefits of ecosystems known as intrinsic values or bequest values that directly enter into the ‘values’ box. Formation of values will be influenced by how robust and accurate the valuation methodologies are in capturing the services from the ecological production functions. For example, decision making criteria, like CBA or the multi criteria method, would depend upon the values arrived through the valuation methods. These decision making criteria would influence the choice and subsequently impact the condition and trend of the ecosystem in consideration. It is very clear that value determines the human choice and the human choice impacts the fate of ecosystems and their services. Valuation of ecosystems thus plays a pivotal role in the designing the appropriate response option.

Valuation of ecosystem services is a growing literature and it is beyond the scheme of this paper to summarise them. But the table below provides a glimpse of the summary:

Table 1 Various valuation methods applied to ecosystem services

Valuation method	Value types	Overview of method	Common types of applications	Examples of ecosystem services valued
Adjusted market prices	Use	Market prices adjusted for distortions such as taxes, subsidies and non-competitive practices.	Food, forest products, R&D benefits.	Crops, livestock, multi-purpose woodland, etc.
Production function methods	Use	Estimation of production functions to isolate the effect of ecosystem services as inputs to the production process.	Environmental impacts on economic activities and livelihoods, including damage costs avoided, due to ecological regulatory and habitat functions	Maintenance of beneficial species; maintenance of arable land and agricultural productivity; support for aquaculture; prevention of damage from erosion and siltation; groundwater recharge; drainage and natural irrigation; storm protection; flood mitigation
Damage cost avoided	Use	Calculates the costs which are avoided by not allowing ecosystem services to degrade.	Storm damage; supplies of clean water; climate change.	Drainage and natural irrigation; storm protection; flood mitigation

Table 1 continued

Valuation method	Value types	Overview of method	Common types of applications	Examples of ecosystem services valued
Averting behavior	Use	Examination of expenditures to avoid damage	Environmental impacts on human health	Pollution control and detoxification
Revealed preference methods	Use	Examine the expenditure made on ecosystem related goods (e.g. travel costs; property prices in low pollution areas).	Recreation; environmental impacts on residential property and human health.	Maintenance of beneficial species, productive ecosystems and biodiversity; storm protection; flood mitigation; air quality, peace and quiet, workplace risk.
Stated preference methods	Use and non-use	Uses surveys to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay for those goods	Recreation; environmental quality, impacts on human health, conservation benefits.	Water quality, species conservation, flood prevention, air quality, and peace and quiet.

Source: Adapted Bateman et al (2010)

In the above Table, the choice of methodology in column 1 is linked with the purpose of valuation and availability of dose response data. In all cases, the information on fundamental ecological production function seems to be absolute necessity. Summarily, while doing valuation following information would make the estimates of ecosystem services as robust, credible and acceptable by decision makers and other stakeholders:

- i. Initial condition of the ecosystem and corresponding ecological production function
- ii. Drivers of change especially the indirect drivers like trade and other macroeconomic factors and its impact on the ecosystem affecting its flow of services
- iii. Units and measurement of ecological services
- iv. Additional perturbances creating changes in flow of ecological services (basically marginal change in ecosystem benefits as a response to marginal change in drivers)
- v. Ecological scale of change and relevant scale of time
- vi. Gainers and losers in the process of change in ecosystem services

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Conserving forest biodiversity and ecosystem services to agriculture

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Abstract

Conservation of forest biodiversity is widely considered to be crucial for sustainable development and society. To increase public motivation to achieve this goal, experts must translate scientific knowledge so that it is understandable to non-scientists. For example, the total forest area in Japan has changed little over the past four decades, leading some to expect that no serious decline of forest biodiversity has occurred. However, several factors have negatively affected biodiversity at the landscape level. The ratio of mature forest has increased because of economic stagnation, including in forestry activities. Natural forests have been converted to plantation forests, but lack of management resources has meant fewer necessary forestry treatments (e.g., longer thinning and harvesting intervals) and abandonment of both plantation and secondary forests. At the landscape level, traditional forest management, which involves clear cutting or selective cutting of secondary forests at short intervals, has declined but is thought to have provided rich pollinator populations and diverse natural enemies to nearby agricultural fields. Social and economic changes have also led to habitat expansion by Sika deer, which destroy natural vegetation through their intensive foraging. Thus, social and economic analyses as well as natural science investigations are essential for developing feasible plans to conserve biodiversity. In terms of forest management, a heterogeneous landscape composed of varied forest vegetation and successional stages is important for providing ecosystem services. To implement sustainable forest management, a system for monitoring ecosystem goods and services should be developed based on collaboration between scientists and non-scientists, including policymakers.

Keywords: forest landscape, forest management ecosystem service flow, secondary forest, sustainability

Introduction

A forest is more than an assemblage of trees. Forests provide habitats for numerous organisms, ranging from microorganisms, algae, and lichens to large plants and animals. Diverse species are directly and indirectly connected to many others in forests, sharing benefits or competing for survival, in niches or through interactions with others. In addition to rich biodiversity, forests also have important ecosystem

functions, such as serving as carbon sinks and playing key roles in nutrient cycling and plant production (Schulze and Mooney 1994). For sustainable use and maintenance of forests and their functions, conservation of ecosystem processes and biodiversity is crucial at both the stand and landscape levels (Lindenmayer et al. 2000).

However, despite the importance of biodiversity to human life, only 30% of respondents to a survey conducted by Japan's Ministry of the Environment in 2004 reported that they had heard about biodiversity (Ministry of the Environment 2007). In 2008, another survey showed that although 83% of Japan's prefectural governments had established numerical targets for conserving biodiversity and the natural environment, only 10% of all local governments, including those in small towns and villages, had set such targets. However, the same survey showed that approximately 98% of prefectures and 30% of total local governments had established goals for global warming (Ministry of the Environment 2008). These results suggest that most Japanese are aware of climate change and will strive to meet climate-related goals within budgetary limits but that there is less concern for biodiversity. However, as shown in a recent declaration on biodiversity by Nippon Keidanren (the Japan Federation of Economic Organizations; Nippon Keidanren 2009), Japanese companies have contributed to biodiversity projects through corporate social responsibility activities. Because companies play an important role in implementing conservation, information sharing among corporate and scientific groups working on biodiversity is becoming more common. Both economic and ecological studies point to the importance of biodiversity not only in terms of ecosystem functions but also in providing ecosystem goods and services to other ecosystems and to society (Daily 1997; Daily et al. 2000; Balmford et al. 2002). With growing awareness of biodiversity, it is important to clarify the connections between biodiversity and ecosystem goods and services so as to devise strategies, plans, and targets for sustainable management and use of natural resources.

In this paper, I examine what kinds of ecosystem services are important for agriculture, how these services are related to forest biodiversity, and what type of forest management is appropriate for maintaining ecosystem service flow from forests to agricultural fields, based on ecological studies. First, I briefly describe trends in forest biodiversity in Japan.

Trends in forest biodiversity in Japan

Japan is a long, narrow archipelago composed of four main islands (Hokkaido, Honshu, Shikoku, and Kyushu from north to south) and more than 6,000 smaller islands. Mountains rising 1,000 to 3,776 m above sea level are located at the center of each main island. The archipelago, which originated from Eurasia, is thought to have separated from the continent approximately 15 million years ago, taking with it some continental flora and fauna. The climate is influenced by the Asian monsoon as well as Japan's location at the eastern edge of the Eurasian continent. Average annual precipitation is 1,700 mm, approximately twice as high as the world average. These geographic, geologic, and climatic characteristics contribute to rich forest biodiversity, including sub-boreal, temperate, sub-tropical, and alpine flora and fauna (National Museum of Nature and Science 2006). Different seasonal durations created by the wide latitudinal range of Japan have also led to the formation of unique species compositions by region (Takayu et al. 2005).

More than 70% of land in Japan is hilly or mountainous and covered with forests. In the past 50 years, the total area of forest cover has changed little. However, change in forest type has occurred. Natural forests have been converted to plantation forests, which now account for 41% of the total forest cover (Fig. 1, Statistics Bureau 2009). The total area of primary forests also increased from 3,764 to 4,591 × 10³ ha between 1990 and 2005 (FAO 2009). Some forest species of mammals, birds, amphibians, reptiles, insects, and vascular plants have become threatened, although few have become extinct (Table 1). The diversity of forest species appears to be stable, although local species compositions and genetic diversity might change significantly with changes in land use.

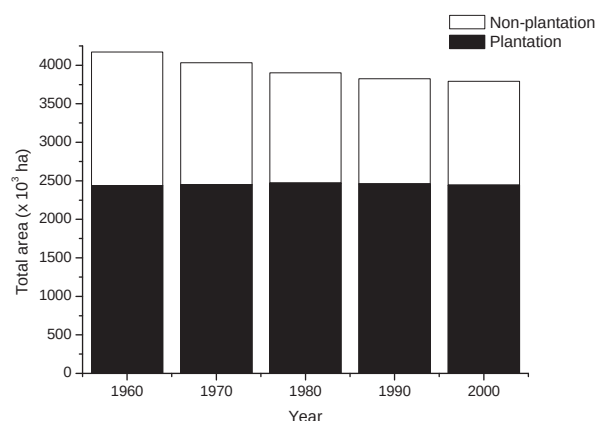


Fig. 1 Change in Japanese forests over the past four decades (Statistics Bureau 2009).

Table 1 Japanese native faunas and flora extinct and under-threat

Taxon	Total species and sub-species no.	Forest dependent species and sub-species	% of extinct in total forest	% of threatened in total forest	Reference ¹⁾
Mammals	152	130	3.1	27.7	A
Birds	540	319	2.2	18.2	B
Amphibians	66	60	0	13.3	C
Reptiles	95	71	0	2.8	C
Butterflies	237	151	0	7.9	D, E
Vascular plants	7,000	1,770 ²⁾	0.5	12.4	A

1) A: Japan Integrated Biodiversity Information System (2010), B: Tojo (2007), C: Okochi (2001), D: Tanaka (1988), E: Inoue (2005).

2) The number of woody plants in total by Kaji (1999).

In light of the 2010 target of the Convention on Biological Diversity, a number of researchers have analyzed trends in forest biodiversity in Japan in recent decades. Since the 1970s, there have been less young forests in Japan, and forest biomass has increased because of stagnation of the forestry industry and changes in society (Forestry Agency 2007; Yamaura et al. 2009a). A survey by the Ministry of Environment showed that forest-dependent bird species did not change in species richness or composition over the 20 years from 1970 to 1990 (Yamaura et al. 2009a). However, bird distributions did change, with distribution areas declining among species that preferred immature forests but expanding among those that preferred mature forests, likely because of the increase in mature forests (Yamaura et al. 2009a). In addition, although migratory bird distributions did not change within Japan, numbers of both immature and mature forest bird species declined, possibly in response to deforestation and consequent loss of habitat in Southeast Asia (Yamaura et al. 2009a). At the same time, the distributions of some medium to large mammals have expanded, probably triggered by reductions in hunting pressure between the 1970s and 1990s (Yamaura and Amano 2010). There was no significant difference in the number of tree species and species composition between the 1990s and 2000s in selected old-growth forests (more than 100 years old) of Japan (Ogawa et al. pers. com.). Because the biomass of such forests is still increasing, they are expected to have not yet reached the climax stage in forest succession (Ogawa et al. pers. com.).

The third National Biodiversity Strategy of Japan (Ministry of the Environment 2007) identified four crises in Japanese biodiversity created by different causes: excessive human activities, insufficient levels of management, introduced alien species, and chemical pollutants and global warming. Researchers have also warned against converting primary and old-growth forests to plantation forests because certain organisms such as epiphytes require large trees for survival. Hattori et al. (2007), for example, found that the average tree diameter at breast height was positively correlated with epiphyte diversity. Invasive

alien species may have different effects on forest flora and fauna. For example, although pine wilt disease caused by the invasive nematode *Bursaphelenchus xylophilus* has affected forest vegetation, the species richness of cerambycid beetles has been unaffected in typical forest landscapes dominated by *Quercus* and *Pinus* trees (Esaki et al. 2005). One of the worst threats to Japanese forest biodiversity today is grazing by Sika deer (*Cervus nippon*). Deer distributions are expanding not only in forests but also into agricultural fields and residential areas probably because of both excessive human activities, insufficient levels of management including habitat loss, less hunting pressure and extra food supply from abundant agricultural field. Intensive deer grazing can devastate vegetation (Uno et al. 2007), and deer-related economic and ecological losses have been reported worldwide (Côte et al. 2004). In the Ohdaigahara region of Japan, expansion of Sika deer populations has had various direct and indirect effects, including reduced dung beetle richness (Sato 2008). Although the threshold of forest biodiversity resilience following deer grazing is unclear, Hino et al. (2003) suggested, based on a system dynamics model, that both Sika deer and sasa bamboo, a preferred food of the deer, should be managed to promote natural regeneration in Ohdaigahara.

Ecosystem goods and services provided by non-plantation forests

Ecosystem goods and services demonstrate the value of biodiversity and can be especially helpful in conveying the importance of conservation to non-scientists, although the correlation between biodiversity and ecosystem services and functions can sometimes seem unclear or redundant (Schulze and Mooney 1994; Daily 1997; Naeem 1998). Forest ecosystem services that are susceptible to biodiversity decline (Dobson et al. 2006) include biological controls (a regulating service) and, pollination and nutrient cycling (supporting services), which support and highly influence agriculture (Swift et al. 2004). As an example, seed and fruit sets are significantly related to the diversity of bees, which are the most important and common agents of crop pollination (Steffan-Dwenter and Tschardtke 1999; Klein et al. 2002a; Ricketts et al. 2004). Bee diversity and seed set are positively related to natural vegetation, including forest vegetation, and negatively correlated with distance from these habitats (Gathmann et al. 1994; Gathmann and Tschardtke 2002; Klein et al. 2002b). Natural enemies such as parasitoids also respond to forests on different scales (Ronald and Taylor 1997). However, as natural vegetation provides a compatible habitat for both potential agricultural pest insects and the natural enemies to control them, integrated habitat management must simultaneously encourage natural enemies and deter pests as much as possible (Landis et al. 2000).

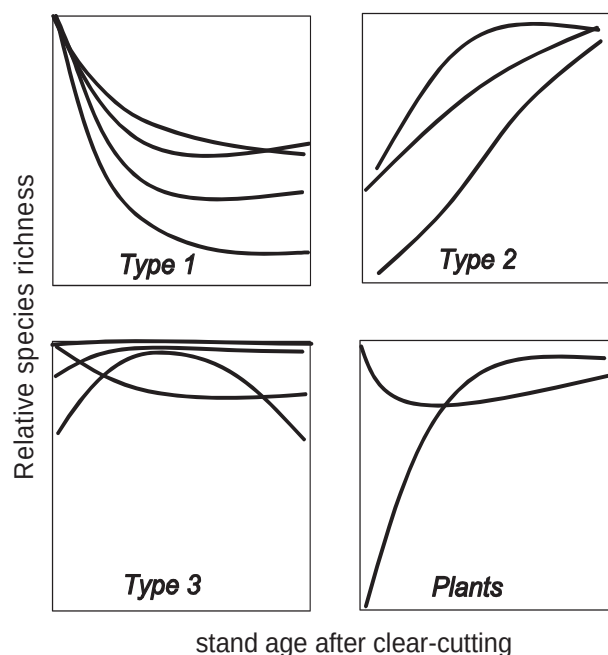


Fig. 2. Trends in the change in species richness after forest clear cutting (Makino et al. 2006).

At the stand level, species richness of insects and mushrooms changes in response to stand age or average tree diameter at breast height after clear cutting in one of three ways: by increasing (mushrooms as decomposers and mushrooms as food for the mite community), decreasing (bees as pollinators, longicorn beetles as decomposers, and tube-renting wasps as predators), or remaining stable (ants and ground beetles as predators and oribatid mites and collembolans as decomposers; Makino et al. 2006; Fig. 2). In particular, bees and parasitoids, the principal ecosystem service agents for agricultural production, are most abundant in clear-cut fields (Malegue et al. 2010; Makino et al. pers. com.). Traditionally in Japan, secondary (i.e., non-plantation) forests near agricultural fields were managed by regular clear cutting and/or selective cutting for fuel. This traditional management method is thought to sustain suitable ecosystem services for agricultural production. Although bees, longicorn beetles, parasitoids, and mushrooms showed richer species diversity in secondary forests than in Japanese cedar (*Cryptomeria japonica*) plantations at the same stand age (Makino et al. 2007; Hattori pers. com.), moths and collembolans differed little in species richness but greatly in species composition between forest types (Hasegawa et al. 2009; Taki et al. 2010b). Thus, quantitatively and perhaps qualitatively, the different pollinators and natural enemies in a landscape composed of different forest stand types are expected to provide various ecosystem services to nearby agricultural fields.

The nutrient cycle is a fundamental ecosystem service that is mostly generated by microorganisms. However, suitable techniques for investigating and quantifying microorganism diversity and services are

still lacking.

Forest management and monitoring to conserve the biodiversity of ecosystem service agents

Harvesting trees in natural forests is a human disturbance that can impact the diversity of forest insects. Spagarino et al. (2001) reported that many functional insect species, including those that benefit agriculture, declined during forest management cycles in *Nothofagus* forests. Although plantations generally lead to less local biodiversity, they can be managed to increase biodiversity compared to the relatively low levels normally found in plantations (Brockerhoff et al. 2008). Conventional forest management techniques in Japan, such as regular thinning of Japanese cedar, also increase the number of certain functional species, such as flying insects (Taki et al. 2010a). Removing competing vegetation that has become overabundant, possibly because of human activities, is a sound measure for promoting forest regeneration. In turn, forest regeneration has been directly linked to biodiversity conservation and hence to the protection of ecosystem services and goods (Nakashizuka 1988; Wada 1993; Takahashi et al. 2007). Because coarse woody debris, snags, and large trees promote additional biodiversity in managed forests as materials for nutrient cycling and habitats (Hansen et al. 1991; Takahashi et al. 2000), forest managers should consider maintaining these resources. Quantification of thresholds at which ecosystem resistance and resilience are overcome is also needed. However, few studies have analyzed such thresholds, despite their importance (Fischer et al. 2006; Thompson et al. 2009). Bird responses to patchy natural forests suggest an adequate statistical approach for determining thresholds regarding necessary conservation areas (Yamaura et al. 2010b).

In Japan, agricultural fields are generally patchily distributed in forested landscapes, and continuous cropping rarely occurs except in orchards. Because of this, it is possible to control the biotic environments around each patch and analyze their effects, such as for the flow of ecosystem services from natural vegetation to an agricultural field. A landscape composed of heterogeneous ecosystems is expected to produce richer species numbers and ecosystem services (Makino et al. 2006; Lindenmayer et al. 2000). Information at the landscape level, for different vegetation types, and at the agricultural patch scale is necessary for assessing ecosystem services to agriculture because the intensity of agriculture and the history of land use are also key factors in species richness (Kremen et al. 2002; Koyanagi et al. 2009).

Biodiversity monitoring has been conducted and evaluated for years at various scales and for various ecosystems (Pereira and Cooper 2006; Baillie et al. 2008). However, the quality and quantity of ecosystem services have rarely been monitored. The lack of

monitoring can be explained by the lack of sufficient surrogates of ecosystem services. In contrast, the number of species has been widely accepted and used as a biodiversity indicator. However, it is impossible to know all species living in a particular monitoring site. As a solution, some non-scientists may view certain umbrella or flagship species as indicators of conservation achievements and scientists may consider keystone species to be better indicators for conservation of biodiversity (Simberloff 1998; Lindenmayer 1999). For monitoring of ecosystem services, the ecological attributes of ecosystems or functional group(s) are better surrogates for analyzing trends instead of using a key species in an ecosystem. Thus, functional diversity, as examined by Flynn et al. (2009), is a potential numerical indicator of ecosystem services.

Conclusions

Over the past four decades, Japanese forest biodiversity has changed little in terms of species numbers, but species composition and biomass have changed. These trends are partly attributable to changes in forest management influenced by socioeconomic changes. To understand trends in biodiversity at scales smaller than the national level, more frequent monitoring (less than 10-year intervals) is recommended to match rapid social, economic, and climate changes.

Assessments of the ecosystem goods and services provided by forests to agricultural production could help motivate non-scientists, including policymakers, to conserve forest biodiversity as well as the forestry industry. Sound monitoring methods for adaptive conservation management should be developed with collaboration between scientists (as the monitoring designers and analyzers) and non-scientists (as the implementing organizations, policymakers, and stakeholders). These methods can include establishing indicators and monitoring bodies.

Conservation of forest biodiversity at the landscape level is essential for conserving the ecosystem service agents that provide regulation and support services to agriculture. Relevant forest management should be proposed to protect both biodiversity and ecosystem services within and flowing from forests.

For implementation of targets, strategies, and action plans, collaboration between scientists and non-scientists, including policymakers, is critical.

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The role of forests in capturing the ecosystem service of pest control: a pathway to integrate pest control and biodiversity conservation

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Abstract

Natural pest control is a valuable ecosystem service that benefits humans in many ways. The presence of perennial non-crop habitats, such as forest remnants, in agricultural landscapes are thought to play a crucial role in maintaining populations of natural enemies of pests and can potentially be managed to provide improved pest suppression in crops and conserve biodiversity. We suggest that a pathway to achieving these targets may include: identification of the key natural enemy species, assessing whether and when natural enemies move between forests and crops, determining whether they can suppress pest population effectively and if so, for how long. When these first several steps are taken successfully, the final step involves a practice transition process, which will determine whether farmers will change their practices as a result of this type of knowledge. We provide several examples from current research that show that natural enemies (as well as some pest species) use forests as reproduction habitat and colonize crops. In addition, we show that pest suppression can be most effective near forest remnants than further away. We conclude with a case study for the implementation of a revegetation program aiming at weed and insect pest suppression. By selecting native plant species that do not support pest species, natural enemy densities can be increased and biodiversity enhanced in degraded agricultural landscapes. Maintaining forest remnants in agricultural landscapes or revegetating weedy degraded land has the potential to provide the private benefit of pest control and the public benefit of biodiversity conservation.

Keywords: forest remnants, insect predators, parasitoids, revegetation by design

Introduction

Forests can provide a range of ecosystem services (i.e. products of nature that yield human well-being; Banzhaf and Boyd 2005). Here we will focus on the ecosystem service of natural pest control in crops, which entails the suppression of pest populations by endemic natural enemies. Because natural enemies can effectively suppress populations of a wide range of insect herbivores, the vast majority do not reach

outbreak levels in crops. The value of this pest control service is estimated at more than US\$ 400 billion per year worldwide (Costanza *et al.* 1997). However there are many agricultural practices that can compromise this ecosystem service, including extensive clearing of perennial habitat and heavy use of insecticides. The presence of perennial non-crop habitats, such as forest (i.e. native perennial woody vegetation) and forest remnants (i.e. patches of remaining native forests often found in heavily cleared agricultural landscapes), are thought to play a crucial role in maintaining populations of natural enemies of pests in agricultural landscapes because they can provide refuge from insecticide, shelter, floral nectar resources, alternative hosts and prey (Bianchi *et al.* 2006).

The ecosystem service of pest control benefits humans in numerous ways, but primarily through reduced pesticide use. This can benefit primary producers by: 1) lowering input costs, 2) reducing exposure to pesticides by applicators, 3) reducing mortality to non-target organisms, such as insect predators which provide pest control services, and 4) minimizing the development of insecticide resistance. This can benefit consumers by: 1) reducing costs of agricultural produce, 2) enhancing food safety, and 3) maintaining food security. This can benefit the environment by reducing harm to non-target organisms, and in turn maintaining biodiversity.

The question that comes to mind is, "How can we better manage for this ecosystem service?" First, we'll need to identify our goal, which – for the purpose of this paper – will be defined as "improved pest suppression by natural enemies and avoid further biodiversity decline". Second, we need to identify and fill in knowledge gaps using principles that can be generalized across agricultural production systems. Third, we need to measure progress. While there are several possible pathways towards enhancing the ecosystem service of pest control, we will here outline one possible pathway. The steps of this pathway include: (i) identifying key natural enemy species providing natural pest control, (ii) assessing whether natural enemies move between forest or forest remnants and crops, (iii) assessing the time of crop colonization, (iv) determining whether they can suppress pest population effectively, and (v) assessing whether they can prolong the time that pest populations are below economic threshold levels.

When these first five steps are taken successfully, the final step (vi) involves a practice transition process, which will determine whether farmers will change their practices as a result of this type of knowledge. For instance, will farmers spray later and less often? Will they maintain and manage their forest remnants? And will they revegetate weedy degraded areas with native perennial vegetation?

Step 1: Identifying key natural enemy species

Pest control providers include numerous species of natural enemies such as arthropods (e.g. herbivores that feed on weeds, predatory arthropods and parasitoids), bacteria, viruses and fungi. Arthropod herbivores can suppress plant growth by boring in stems, and feeding on roots, leaves, flowers and fruits (Debach and Rosen 1991). Predatory arthropods comprise a wide range of orders and kill and consume their prey (Debach and Rosen 1991). Parasitoids include species of Diptera (flies) and Hymenoptera (wasps) that oviposit (lay their eggs) externally on, or internally in their host. Parasitoid larvae feed on host tissues and kill their hosts before developing into adults. Some bacteria kill and consume other arthropods once they are ingested by an insect. The best known and now widely commercialized example is the bacterium *Bacillus thuringiensis* or Bt, which is used to control a wide range of lepidopteran (caterpillar and moth) pests of agricultural crops. Viruses and fungi can infect and potentially kill their hosts. However, the focus of this paper will be on arthropod predators and parasitoids of agricultural pests.

Natural enemies can be either native or exotic (i.e. introduced). When an introduced plant becomes a weed or an introduced arthropod becomes a pest, natural enemies may be needed to be introduced to help control the pest. This is referred to as classical biological control (Debach and Rosen 1991). When a native plant or insect becomes a pest, natural enemies may need to be managed to increase their effectiveness at controlling the pest, which is referred to as conservation biological control. Regardless of whether the pests are exotic or native, practices to conserve and manage natural enemies will be a key component of capturing the ecosystem service of pest control.

In cropping systems where integrated pest management (IPM) is practiced the key pests and natural enemies are usually known. Many pest species are causing damage in many crop types across different industries (Schellhorn *et al.* 2008). For example, the cotton bollworm and native bud worm, *Helicoverpa armigera* and *H. punctigera* (Lepidoptera: Noctuidae), respectively, are pests of cotton, many grain crops (sorghum, sunflower, lucerne), as well as horticultural crops (sweet corn, tomatoes, fresh beans). The diamondback moth, *Plutella xylostella*, is a pest of Brassica vegetables and

a pest of canola. The same is true for many natural enemies whereby often the same species provide pest control in different crop types.

Step2: Assessing whether natural enemies move between forests, forest remnants and crops

In Australia, several studies have documented the occurrence of predators and parasitoids of agricultural pests in native perennial vegetation (e.g. Rencken 2006, Stephens *et al.* 2006). Immature stages of insect predators have been found on a range of native plants indicating that these habitats are used for reproduction, so-called source habitats (Schellhorn *et al.* 2008). Several types of natural enemies, such as brown lacewings and ladybird beetles, move between habitats at distances greater than a few hundred meters (Schellhorn and Silberbauer 2002, Silberbauer *et al.* 2004) or even further than 1 km in 2 days (Schellhorn unpublished). Natural enemies can also move at regional scales, such as green lacewings (Neuroptera: Chrysopidae), a key predator of aphids, that migrates over distances up to 300 km in the UK (Chapman *et al.* 2006), even though this has hardly been documented. Although there is a considerable number of studies that suggest that natural enemies are influenced by non-crop habitats including forests (Bianchi *et al.* 2006), there is limited information on natural enemy dispersal and migration, with the exception of ground-dwelling carabids (Coleoptera: Carabidae) (e.g. Dennis and Fry 1992).

To fill in this knowledge gap, we quantified movement patterns of natural enemies within and between forests, forest remnants and crops using bi-directional interception traps in a major vegetable production region of southeast QLD, Australia. Two habitat edges (i.e. crop-forest remnant edges), and two habitats (i.e. forest and crop) were evaluated. The edges differed in that one was a riparian (i.e. watercourse habitat) remnant forest-crop edge (RF-Crop), and the other was a forest-crop edge (F-C). After trapping for one year (May 2007-08) and focusing on 15 insect predator and 1 herbivore species (Table 1), four key findings emerged: 1) all species occupied all four habitats, but the forest interior had the lowest activity density (Fig. 1), 2) there is strong species-specific preference for some habitats (e.g. *D. bellulus* and *E. vividaureus* had a significantly higher activity density in the RF edge than in the crop, whereas for *S. macrogaster*, this was the reverse). 3) there is significantly more net immigration of natural enemies from RF into the crop than vice versa, which is also true for the leafhopper pest, *Cicadulina bimaculata*, and 4) a diversity of forest habitat is important to support a diversity of pest control agents. For example, Robber flies, Asilidae spp., had significantly higher activity densities in the F-C edge than in other habitats ($X^2=20.54$, $P<0.0001$), and had higher net immigration from the forest into the crop

Table 1 The focal species of predators and pests used in the example

Status	Order	Family	Scientific Name	Common Name
Predators	Coleoptera	Coccinellidae	<i>Diomus notescens</i>	Minute two-spotted ladybeetle
	Coleoptera	Coccinellidae	<i>Coelophora inaequalis</i>	Variable ladybeetle
	Coleoptera	Coccinellidae	<i>Hippodamia variegata</i>	White collared ladybeetle
	Coleoptera	Coccinellidae	<i>Coccinella transversalis</i>	Transverse ladybeetle
	Coleoptera	Coccinellidae	<i>Micraspis frenata</i>	Striped ladybeetle
	Coleoptera	Coccinellidae	<i>Harmonia conformis</i>	Common spotted ladybeetle
	Coleoptera	Melyridae	<i>Dicranolaius bellulus</i>	Red & blue beetle
	Diptera	Asilidae	<i>Morpho spp78</i>	Robber fly
	Diptera	Asilidae	<i>Morpho spp79</i>	Robber fly
	Diptera	Syrphidae	<i>Melangyna (Austrosyrphus) sp.</i>	Hoverfly
	Diptera	Syrphidae	<i>Sphaerophoria macrogaster</i>	Hoverfly
	Diptera	Syrphidae	<i>Simosyrphus grandicornis</i>	Hoverfly
	Diptera	Syrphidae	<i>Episyrphus viridaureus</i>	Hoverfly
	Neuroptera	Chrysopidae	<i>Mallada spp.</i>	Green lacewing
	Neuroptera	Hemerobiidae	<i>Micromus spp.</i>	Brown lacewing
Pest	Hemiptera	Cicadellidae	<i>Cicadulina bimaculata</i>	Maize leafhopper

($F_{1,3}=29.6$, $P<0.0001$), than vice versa. These data highlight that there are strong links between forest, forest remnants and crops, even though the actual function of each of these habitats is still unknown.

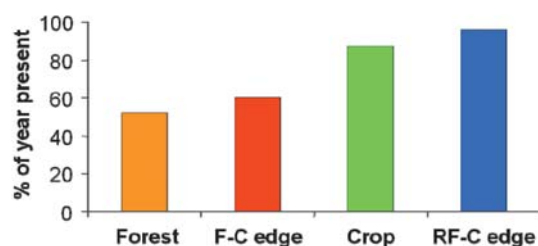


Fig. 1 The percent of the year each habitat is occupied by the 15 focal predators in the landscape with a high (>70%) percent cover of native vegetation.

Step 3: Assessing the time to crop colonization

Theoretical and empirical studies have highlighted the importance of early arrival (immigration) and colonization of natural enemies for effective pest control (Settle *et al.* 1996; Ives and Settle 1997; Landis and van der Werf 1997; Bianchi and van der Werf 2003). Mortality of pests at an early stage when their populations are increasing can prevent a potentially large number of offspring in future

generations (van der Werf 1995). The density and timing of arrival of natural enemies to crops is dependent upon (among other things) their dispersal ability, the distance to source populations of immigrants and the way in which they aggregate to prey (Corbett and Plant 1993; Corbett and Rosenheim 1996; Schellhorn and Andow 2005). Therefore, the composition of the surrounding landscape can greatly influence the number of immigrating natural enemies in crops as it determines the distances that natural enemies must travel from their source (reproduction) habitats to newly planted crops (Schellhorn *et al.* 2008; Bianchi *et al.* 2009).

The landscape is comprised of many habitat types (e.g. crops, grazing land and forests), so understanding the function of these habitats and how they change over time allow us to identify features of pest suppressive landscapes (Bianchi *et al.* submitted). Forests, which are typically the main perennial non-crop habitat, are relatively constant in space and time, while crops are changing due to harvesting and crop rotation. There is also a diversity of boundaries or edges in these agricultural landscapes, such as crop-forest and crop-grazing land edges. Insects are known to respond to these edges (Fagan *et al.* 1999, Olson and Andow 2008), both in terms of their community composition and how they interact between adjoining patches. Although early colonization of natural enemies is considered an important factor for effective pest

suppression, and several studies have demonstrated the benefits of forest habitats in agricultural landscapes for harnessing the ecosystem service of pest control, very little attention has been given to the question ‘Do crops adjacent to forest benefit from earlier immigration and colonization of natural enemies than crops far from forests?’ Honek (1982) documented the colonization of the ladybeetle *Propylaea quatuordecimpunctata* of crops from forest edges in early spring. He showed that *P. quatuordecimpunctata* was slightly earlier in fields close to forest edges, but ladybeetles effectively colonized a distance of 2 km from forests in 3–4 days. However, more studies are needed to document the process of crop colonization.

Step 4: Determining whether natural enemies can suppress pest population effectively

Several studies have shown correlations between the amount of non-crop habitat in agricultural landscapes and densities of natural enemies and pests (see review by Bianchi *et al.* 2006). These studies typically correlate response variables such as the amount of herbivory or the percent of parasitism to physical metrics of the landscape such as the percentage of non-crop habitat, (e.g. hedge rows, forests). The analysis shows that in 74% and 45% of studies reviewed, respectively, natural enemy populations were higher and pest populations lower in landscapes with a high percentage of non-crop habitats versus low percentage of non-crop habitat. Although these studies have clearly demonstrated a link between non-crop habitat and pest control, due to the correlative nature of the studies the underlying mechanisms remain unknown, which makes it difficult to identify effective strategies to manage for enhanced pest control services while maintaining biodiversity. Understanding the function of habitats in terms of source and sink habitats for pests and natural enemies and how this function changes in space and time in the agricultural landscape mosaic is a promising strategy to reveal mechanisms underlying the ecosystem service of pest control and formulate effective management strategies to manage for this service.

To get a better mechanistic understanding of the role of forests in harnessing the ecosystem service of pest control, we conducted an experiment to quantify how the distance to native forests remnants influences crop colonization and the potential of natural enemies to suppress pest populations. We used a pest-parasitoid system, silverleaf whitefly (SLW) *Bemisia tabaci* (Hemiptera: Sternorrhyncha: Aleyrodidae) and its parasitoids (*Encarsia* spp and *Eretmocerus* spp) in two cotton/grain landscapes in the Darling Downs (southeastern Queensland, Australia) with 6 and 13% native forest remnant. In each of these landscapes we established experimental fields (60m x 80m) by placing sentinel cotton seedlings infested with SLW

nymphs in native forest remnants, on fallow fields (e.g. bare soil) adjacent to remnants, or fallow fields >400m from remnants. Two key findings emerged. Firstly, parasitism of SLW nymphs was the highest in remnants, followed by fields adjacent to forest remnant, and was virtually absent in fields far from remnant suggesting that the source of parasitoids is the forest remnant from which they colonize the surrounding fields. Secondly, although SLW parasitism was clearly associated with forest remnants, it was the landscape with 6% forest in which the highest parasitism rates were recorded, suggesting that local characteristics of forest remnants can play a crucial role for supporting parasitoid populations. To get a better insight of the habitat use of pests and natural enemies, we also sampled crops and native remnant forests. Using the high-forest (13 %) landscape to illustrate an example, we found a higher predator to pest ratio in five out of six native forest sites as compared to one out of six in crops (Bianchi *et al.* submitted). This shows that natural enemies and pests are typically found in both forest remnants and crops, but in very different ratios.

Step 5: Assessing whether natural enemies can prolong the time that pest populations are below economic threshold levels

As mentioned earlier, early arrival of natural enemies relative to the pest is expected to be a key determinant for effective pest suppression. However, there are very few studies that link any type of non-crop vegetation, the time to colonization of natural enemies and the period that pest densities are below economic threshold levels. Galecka (1966) demonstrated that in potato fields near forests aphid predators (including Coccinellidae, Syrphidae, Araneae, Chrysopidae and Anthocoridae) arrived earlier and in higher numbers than in potato fields far from forests. As a consequence, the rate of increase in aphid populations was lower in fields near forests than in fields further away from forests. More work is needed on this topic.

Step 6: Implementation of sustainable practices

Strategies aiming to implement sustainable practices pose technical, biological, and even more importantly, social challenges (Schellhorn *et al.* 2008). Here we will provide an example of the implementation of a sustainable practice for revegetating weedy degraded on-farm areas with native perennial vegetation, which may also assist with reducing agricultural pests and disease problems. The work was conducted in a landscape characterized by intensive managed vegetable crops, non-productive areas supporting exotic weedy species, and little remaining native vegetation (forest or otherwise) on the Northern Adelaide Plains in South Australia (Australia). In these cropping systems numerous species of weeds are

known to harbor pests and diseases of vegetable crops. Weed control is often costly, providing only short-term relief, and causes environmental problems such as erosion, excessive dust, and poor soil moisture. Although there is incentive for controlling weeds in cropping areas because they can compete directly with the crop, there is less incentive for controlling weeds along drainage ditches, fence rows, and land surrounding fields.

The work involved the integration of Australian native perennial woody vegetation with vegetable production, with a focus on replacing weeds that host insect pests and diseases with native plants that do not, ultimately manipulating vegetation to disadvantage pests and disease at a farm scale (Schellhorn *et al.* 2009, 2010). Focusing on three exotic and one native species of pest thrips (Western Flower thrips *Franklinella occidentalis*, tomato thrips *F. schultzei*, onion thrips *Thrips tabaci*, and plague thrips *T. imaginis*, respectively), we determined their occupancy and density on 31 plant taxa (19 exotic and 12 native) at weekly intervals during a whole year. We found that weeds in the plant families Brassicaceae and Solanaceae supported pest thrips more often and in higher densities than native plants. For example, *F. occidentalis* was eight times more likely to be found on solanaceous weeds than other plant taxa. In contrast, they were hardly encountered on native plants belonging to Myrtaceae and Chenopodiaceae. We also showed that: 1) the native plants (but also weeds) are important reservoirs for natural enemies, 2) revegetation is 2.5 times cheaper than the bare-earth approach, and 3) harvesting the seeds from the native plants provides extra income when sold to the native vegetation propagation industry. The revegetation approach could potentially provide an opportunity for thrips management with other potential benefits such as reduced long-term management of weeds and revegetated areas, minimal top-soil erosion, increased biodiversity, increased public amenity/aesthetic values and most importantly, improved sustainability of horticulture (Stephens *et al.* 2006, Schellhorn *et al.* 2009; 2010)

If crops adjacent to forest benefit from earlier colonization of natural enemies and keep pest populations below threshold for longer, then in theory (assuming farmers spray only when pest exceed a threshold) less insecticide will be applied. Delayed application of insecticide also enables farmers to rely on pest control from natural enemies for longer. Although there is considerable interest in establishing field margins, flower strips and beetle banks, there is much less known how to manage native perennial vegetation, and whether this can influence the timing of crop arrival and colonization by natural enemies, reduce the rate of population growth of pest populations in crops, and farmers' decision to hold-off spraying. Successful examples of this type may provide incentive for farmers to maintain and manage forest and forest remnants. Forest remnants acting as

sources for natural enemies may not only support pest control in nearby crops, but can also enhance the ecosystem service of pest control at larger spatial scales by supplying natural enemies to the 'global' population or meta-population of natural enemies (Levins 1969; Bianchi unpublished data). Whether the contribution to the global population is enough incentive for growers to change practice remains to be seen. It is a positive signal that many growers have embraced the concept of replacing weeds with native perennials around their glasshouses. However, there is a clear economic benefit in weed control alone.

Conclusion

Non-crop vegetation, particularly forest, forest remnants and native perennial woody vegetation can play a significant role in capturing the private benefit of pest control and the public benefit of biodiversity conservation. However, as demonstrated, in some circumstances non-crop vegetation can function as a source of pests that colonize crops, (e.g. the example above of exotic grass growing in the transition zone between the crop and forest hosting the jassid, which then moves into the crop.) This result provides a caution for generalizing non-crop habitat as only a pest-control benefit, and underscores that sometimes there will be trade-offs. The question then becomes whether we can manage the situation to minimize the negative result.

Meeting the target of pest control and biodiversity conservation is challenging and we propose to adopt a pathway based on a scientific framework which allows us to understand habitat function in space and time for natural enemies and pests. If our results demonstrate incentive for land owners to change current land management, the bigger challenge will be achieving a change in practice.

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Importance of diversity in foods and culture for sustainable resource use

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Some may consider a negative relationship between level of resource use and biodiversity. However, Japan National Strategy for Biodiversity and Action Plan (NSBAP) wrote that Japanese biodiversity is threatened by underuse by human. According to Millennium Ecosystem Assessment (MA 2005), there are 5 major direct driving forces that may harm ecosystem services; (A) habitat change, (B) climate change, (C) invasive species, (D) over-exploitation and (E) pollution. These factors are driven by 5 “indirect driving forces”; demographic, economic, sociopolitical, cultural/religious and science-technological factors. Direct driving forces harms ecosystem services and human well-being. Human well-being is also harmed by loss of ecosystem services. In Japanese Sub-global Millennium Ecosystem Assessment (J-SGA), we added under-use as the 6th direct drivers (Fig. 1).

Japanese forests are still rich, and the volume of woods is gradually increasing. However, Japanese forest products has been decreasing since 1960s. Japanese human well-being depends on imported woods. Because of globalization in bioresource use, human well-being in a local society does not depend on the local ecosystem services, irrespective of degradation of local ecosystem functions or biomass. We feel some ambiguity of the definition of ecosystem service, productivity from ecosystems or standing bioresource and functions in the ecosystem. Therefore J-SGA shows both, as shown in Table 1. In

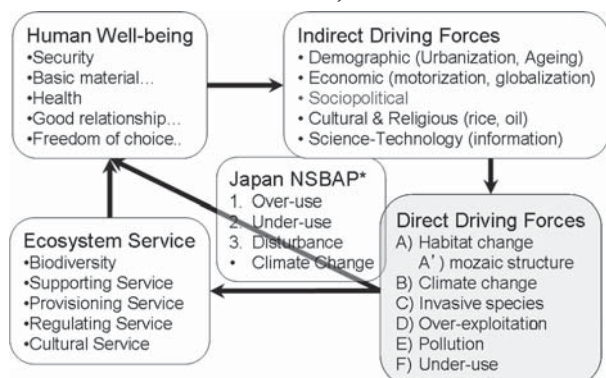


Fig. 1. Scheme of Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005) and its modification of Japan Sub-global Assessment (United Nations University 2010). NSBAP means National Strategy for Biodiversity and Action Plan.

Table 1. Trends in ecosystem services in Japan (United Nations University 2010)

Ecosystem services	Flux	Potential	Notes
Provisioning service	Foods	↘ →	increase imports and stock products
	woods/fibers	↘ ↗	Import woods, decrease fibers potential
Regulating service	Air	± ±	clean-up by abandoned paddy fields
	Water	± ±	degradation of water quality in 50 yrs
	Soil	± ±	
	living things	↘ ↘	serious pollen disease by cedar forests
Cultural service	Spiritual	↘	decrease shrine/temple forests
	recreational	→	decrease hiking/ increase ecotourism
	Arts	↘	decrease crafts
Supporting service	Terrestrial	↘	wetlands, rivers, plairies
	marine	↘	tidal flood, beach,
Biodiversity		↘	decrease of rare & common species

Japan, foods imports from abroad but stock farm potential has increased. Productivity of paddy field per unit area increased but the area of paddy field decreased due to abandonment in Satoyama regions (United Nations University 2010). Biomass energy in developed countries has been replaced by fossil fuel, despite the fact that woods are not yet exhausted. In addition, experience from “Satoyama” and “high nature value farmland” in Europe shows that underuse can actually be a problem for biodiversity.

Provisioning services are supported by primary production and material cycling (supporting services), which is probably enhanced by biodiversity. The reason why biodiversity enhance ecosystem services is usually explained by a similar idea of the classic theory, diversity-stability hypothesis (Elton 1958). The diversity-stability hypothesis means that community with higher diversity is usually more stable and robust against disturbance. Although this intuitive hypothesis is convincing, the mathematical evidence has not long been obtained (May 1972). Recently, our recognition of stability of communities

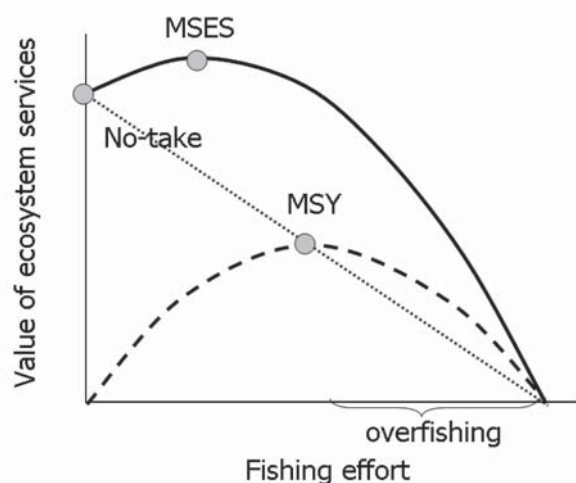


Fig. 2. Scheme of paradigm shift from overfishing, MSY (maximum sustainable yield), no-take and MSES (maximum sustainable ecosystem service). Bold, broken and dotted lines mean the values of total ecosystem service, ecosystem service other than fisheries and fisheries yield, respectively.

has slightly changed. The total biomass of a community with rich species diversity is stable, although biomass of each species may change (Tilman 1999).

Classical fisheries theory recommended sustainable fisheries that maximize the long-term fisheries yield. This idea is called maximum sustainable yield (MSY). However, the fisheries yield is a small part of ecosystem services. Fisheries yield increases with catch amount. The catch amount is the product of fishing effort and standing biomass of the target bioresource. The stock abundance usually

decreases as the fishing effort increases. According to a simple mathematical model, the catch amount is a hyperbolic function of fishing effort. The value of total ecosystem services is maximized when the fishing effort is smaller than that for MSY (Fig. 2). Matsuda et al. (2008) recommended a new idea of the maximum sustainable ecosystem service (MSES) rather than MSY.

Ecosystem is characterized by uncertainty, non-equilibrium and complexity in species interactions. The classic idea of MSY does not take into account of anything of these (Matsuda & Abrams 2006). MSY is based on perfect knowledge of the relationship between standing biomass and reproduction, equilibrium theory and single species dynamics. Adaptive management is robust against uncertainty and non-equilibrium dynamics. We also need ecosystem approach in biodiversity conservation and fisheries.

This suggests that resource management of single species is hopeless, the resource abundance may fluctuate even without harvesting. Deer and sardine are typical examples of such natural fluctuating resources. Therefore, we need to build resource management of multiple species, or rather ecosystem-based resource management (Matsuda and Katsukawa 2002).

Marine food web is usually complicated, starting from phytoplankton, sea weed and algae, and detritus. Small fish such as anchovy usually eats zooplankton. Squid eats fish such as anchovy. And toothed whales eat anchovy and squid, therefore the trophic level of shark and toothed whales are approximately 5 (Fig. 3). There are several taxa that are not eaten in the 1st and

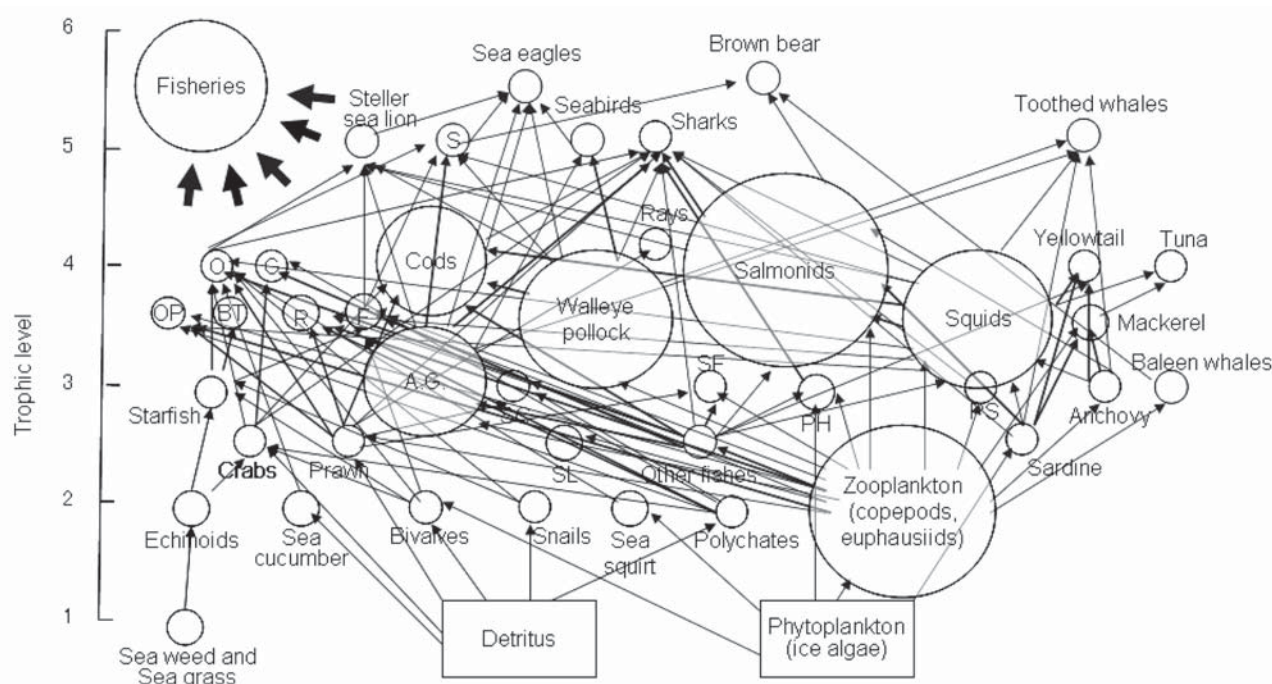


Fig. 3. Marine food web of the Shiretoko area as depicted by the Scientific Council of Shiretoko World Natural Heritage. It includes bears and sea eagles. Grey circles represent taxa that are used by fishers or human, and those catch statistics are compiled.

2nd trophic levels, phytoplankton, zooplankton, detritus, sea cucumber and starfish. Japanese eat many other fish and marine organisms and fishers compile the catch amount and yield of these taxa. If catch amount or the ratio of yield to catch has substantially decreased, the fisheries may not be sustainable because the fish price usually decreases with fish size. Information compiled by fishers is useful to monitor ecosystem status. The fish size and age often decrease by overfishing. In Shiretoko World Heritage site in Japan, catch of sardine, anchovy, red king crab, *Sebastes* and herring substantially decreased by >96%. Greenling decreased their catch by 70% and the fish price by 64% (Matsuda et al. 2009).

There are several merits to harvest multiple species: (1) stabilizing the total yield from the community rather stable than the case of harvesting a single species, (2) reducing monitoring cost of unused species, (3) reducing exploitation rate when the resource abundance is low. The target switching in fisheries is effective on multispecies fisheries management (Matsuda and Katsukawa 2002). In addition, resource management may be easier than nuisance control of deer (Matsuda et al. 1999). These mean that diversity in foods and culture is important for resource management.

Finally, cultural diversity is as important as biodiversity for sustainable society. Environmental standard sometimes seek zero or minimum risk. The balance between ecological risk and socioeconomic benefit depends on society. For example, the western society who does not eat fish made a too low concentration of mercury for food safety. Rice and fish are important and good food although these are contaminated heavy metals and dioxins of which concentrations exceed European food safety standard. We need a new idea with environmental risks. Paddy field and artisanal fisheries are environment friendly.

Several concepts are used for environment-friendly life, external market value, ecosystem services, health risk, forest/marine stewardship, and food mileage. These ideas have shorter history than a few decades. It is suspected that these ideas may disappear in a few decades future. On the other hand, several traditional or religious concepts have been used to avoid some mistakes (Table 2). The global standard may change from decades to decades, and it is sometimes ambiguous. Rabbit hutch has few ecological footprint but it was used for criticism to Japanese economic activity a decade ago. We usually use the concept of ecosystem service for the reason of nature conservation. Such utilitarianism is often powerful, but we need other reason for nature conservation. And the contents of the latter differ among nations, cultures and people.

Table 2 Dichotomy of western and eastern ideas of environment-friendly life

Western/scientific ideas	Eastern/traditional ideas
External-market value	Mottainai (keep redundancy)
Ecosystem services	Grace of nature
Risk/benefit tradeoff	Moderate
Marine/forest stewardship	Awed by nature
Food mileage	Local foods for local consumption
Public involvement	Mutual consensus
Contract	Regards without explicit requests

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Restoring Biodiversity and Forest Ecosystem Services in Degraded Tropical Landscapes

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Abstract

Over the past century, an estimated 850 million ha of the world's tropical forests have been lost or severely degraded, with serious impacts on local and regional biodiversity. A significant proportion of these lands were originally cleared of their forest cover for agricultural development or other economic uses. Today, however, they provide few if any environmental goods or services to society, and in particular to forest-dependent rural populations. Despite the scientific, social, economic, and policy challenges involved, the rehabilitation of degraded tropical forests and landscapes offers a major opportunity to both improve the livelihoods of the rural poor and to reverse the seemingly relentless tide of biodiversity loss and environmental degradation worldwide. Meeting this challenge requires an understanding of the underlying socio-economic, cultural and political drivers of deforestation and ecosystem degradation at local and broader spatial scales. These insights can be used to inform strategies for biodiversity conservation linked to enhancement of livelihoods, and development of innovative, participatory, approaches to forest restoration that combine the collective knowledge gained through decades of research in the fields of forestry and ecology, and the traditional knowledge of forest-dependent people who have the most to gain or lose in the process. This paper will explore a range of strategies and options developed and used in tropical countries to restore both biodiversity values and other economically and socially important environmental goods and services in degraded tropical forest landscapes.

Keywords: biodiversity, ecological restoration, environmental services, landscapes, tropics

Biodiversity, forest ecosystem services, and human well-being – making the connections

Biodiversity is a cornerstone for the provision of many of the forest ecosystem services upon which all human societies depend (Diaz et al. 2005, Campos et al. 2005, Fischlin et al. 2007, Louman et al 2009). These include supporting services (e.g., nutrient cycling, soil formation), which underpin provisioning services (including wood products, non-wood

products, clean and plentiful water), regulating services (e.g., climate regulation, flood regulation, pest and disease regulation), and cultural services (aesthetic, spiritual, recreational values). All of these contribute to human well-being, providing basic materials for life, health, security, harmonious social relations, and freedom of choice and action with respect to natural resource use.

The growing appreciation of the important role of biodiversity in the provision of ecosystem goods and services could inform and strengthen efforts to better harmonize sustainable economic development and biodiversity conservation objectives. In the context of many tropical forest regions, where the needs for both sustainable economic and social development and biodiversity conservation are perhaps more urgent than in other parts of the world, what options exist for reconciling these seemingly conflicting objectives? How can forested landscapes be managed to increase their biodiversity values, improve the flow of ecosystem goods and services to local communities, and contribute to food security and economic development?

Given the complex patterns of land ownership and use, the legacies of generations of forest and agricultural management, and fragmentation of forest ecosystems typical of most tropical landscapes, there are no simple answers to these questions. Rather, solutions are likely to be found in a variety of existing and innovative agricultural and forest management strategies and practices that, can collectively, at a landscape scale, meet these seemingly conflicting environmental and socio-economic objectives.

In this paper, often overlooked opportunities for biodiversity conservation and enhancement of ecosystem services that exist outside of protected areas will be discussed, with an emphasis on degraded tropical forest landscapes. Such landscapes include extensive areas in which biodiversity, provision of ecosystem services, and economic and social values have been diminished due to excessive use and/or mismanagement to the extent that their capacity to naturally recover their structure and potential ecological functions on a time scale compatible with society's needs (for their goods and services) has been undermined.

A landscape perspective on tropical forest biodiversity conservation

Among the many lessons learned by ecologists and conservation biologists in recent decades is the importance of landscape ecology. Landscape ecology tracks the changes and interconnections among habitats and ecosystems across broad spatial scales and the role of landscape-level ecosystem processes (e.g., hydrological and biogeochemical cycles), that connect ecosystems and influence their structure and function (cf. Hansson and Angelstam 1991, Forman 1995, Menon and Bawa 1997, Gutzwiller KJ 2002). Major disturbances in one component of a landscape can have serious repercussions and far-reaching impacts on biodiversity and ecosystem function elsewhere on the landscape. For example, the degradation or deforestation of tropical montane forests can result in significant disruption of regional hydrology, and negatively impact seasonal streamflow patterns throughout the affected watershed (Ataroff and Rada 2000, Calder 2002). These changes may in turn may affect the dynamics of aquatic and estuarine ecosystems and diminish the quality of the watershed as habitat for freshwater and anadromous fish and invertebrate populations, and their availability to people who rely on these resources for their livelihoods and food security.

Another important feature of forested landscapes, particularly human-dominated ones, is their dynamic nature. Natural disturbances such as fire, extreme weather events (storms, drought, severe winds), and major pest outbreaks frequently alter the structure and function of forest stands at varying scales, creating a shifting mosaic of habitats and successional stages across the landscape. Changes in the legal and political context – the laws, institutions, and governance structures affecting land ownership, tenure and management, and the changing economic, social, and cultural needs and aspirations of the people who earn their livelihoods in these landscapes, result in often unpredictable changes in agricultural and other land-use practices that affect the distribution and condition of forests and other ecosystems, as well as biodiversity (of species and habitats) at local and larger spatial scales. A dramatic example of this can be seen in Caribbean island of Puerto Rico, where the extent of natural forest cover has increased, through natural regeneration processes, from less than 10% in the 1940s to over 40% today due to economic changes that produced a dramatic shift in land uses, i.e., abandonment of extensive sugar and coffee production and smallholder agriculture (Grau et al 2003).

Landscapes in the tropics, as elsewhere, are increasingly fragmented. They are comprised of a variety of natural, semi-natural, and intensively managed (including agricultural) ecosystems subject to varying degrees of human management, under a variety of individual, collective or state ownership and

governance systems. They support increasing numbers of people whose depend to varying degrees, directly or indirectly, on the environmental goods and services that these landscapes and their component ecosystems provide. This reality, needless to say, presents significant challenges for effective land-use and biodiversity conservation planning. In the absence of “enlightened” authoritarian control over conservation and land-use decision-making and management, balancing the competing political, economic and social interests and perspectives of stakeholders is a formidable task. Thus, biological diversity conservation objectives are often compromised in favor of more immediate political concerns or economic development objectives.

Our understanding of the dynamic nature of landscapes, of which humans must be considered an integral part, and our appreciation of the impacts of both natural disturbance regimes and shifting demands for ecosystem goods and services on biodiversity, requires a shift in conventional thinking about conservation and sustainable use of biodiversity and other natural resources (Wallington et al 2005). It is becoming increasingly clear that while establishment and management of protected area networks are of fundamental importance for biodiversity conservation, there are limits to their effectiveness, particularly in tropical landscapes. Here, increasing demands for agricultural land, water resources, energy, timber, and non-timber forest resources pose serious challenges for sustainable management of protected areas to meet environmental, economic and social needs. As conservationists and land managers worldwide have learned, focusing exclusively on the biophysical components and ecological dynamics of protected areas or other land management units, while ignoring (or giving only scant attention to) the human dimensions of landscapes in which conservation management units exist, can lead to conflicts that undermine the effectiveness of biodiversity conservation and the provision of environmental goods and services that these ecosystems provide. Furthermore, an exclusive focus on protected areas may overlook important opportunities for enhancing biodiversity conservation in relatively undisturbed and even intensively managed habitats.

A landscape perspective, useful as it is for enhancing our understanding of the inter-relations between habitats and ecosystems and their importance for biodiversity conservation, is invaluable for other reasons. It is essential for the development of politically, economically, and socially viable, and effective, approaches for sustainable development that include the conservation and sustainable use of biological resources. This reality has been recognized by the Convention on Biological Diversity in its promotion of the “ecosystem approach” as a framework for the integrated management of land, water and living resources to promote conservation and sustainable use of biological diversity. Applied to

forests and forested landscapes, the ecosystem approach is very similar to, and compatible with, the concept of sustainable forest management¹ (SFM) used within the forestry community and international forest policy forums such as the United Nations Forum on Forests. Both approaches recognize the need for balance and harmonization of environmental, social and economic objectives in the development of forest policies and management strategies and practices.

Strategies for enhancing biodiversity and ecosystem services in complex human-dominated tropical landscapes

Understanding the context and establishing priorities

As discussed above, the limitations and opportunities for enhancing biodiversity conservation and sustaining provision of ecosystems goods and services in complex tropical landscapes require an appreciation of certain realities. First, that change is a constant within ecosystems and the human societies that exist and are sustained within them. Second, that historical, political, legal, economic, social and cultural contexts determine the ways in which individuals and communities manage their lands to meet their food security and livelihood needs. Third, that biodiversity conservation (and ecosystem service) objectives must be compatible with the legitimate, and variable, needs and aspirations of stakeholders. This implies the need for a very flexible approach in promoting biodiversity conservation (or restoration) at the landscape level, one that recognizes the biophysical and socioeconomic limitations (but also the opportunities) for biodiversity conservation (or its enhancement) under different land-uses and natural forest conditions (Sayer et al 2004, Michon et al 2007). This raises a number of issues that require “negotiations” among stakeholders – individual landowners, communities, state land management authorities including those concerned with biodiversity conservation, and in some cases conservation and development NGOs, to develop viable strategies that will optimize biodiversity conservation and provision of biodiversity-mediated ecosystem services at the landscape scale.

Of particular importance is prioritization of biodiversity conservation and ecosystem services objectives. Since different land management practices will produce different outcomes in terms of

biodiversity and provision of ecosystem services, there is a need to determine which ecosystem services and which components of biodiversity are most important, both at the individual land management (e.g., farm, forest stand) unit level, but also at the broader watershed or landscape level. These are societal choices which will inevitably involve trade-offs, at least at the local (site) level, as management for one or a limited number of ecosystem services may reduce provision of other services (e.g., carbon storage vs provision of non-timber forest products). Similarly, managing primarily for conservation of a limited subset of plant or animal species or species assemblages may have detrimental impacts on other biodiversity components with very different habitat requirements. While these trade-offs are inevitable, at least at the site level, having a common agreement among stakeholders of the objectives at the landscape level and understanding (and communicating) the influence of different land/habitat management practices on different components of biodiversity and effects on provision of different ecosystem services can help to optimize attainment of the principal biodiversity and ecosystem service provision goals at the broader, collective (landscape) level.

A second area of concern, and even greater uncertainty, is the cumulative effect at the landscape level of applying different management practices designed for particular desired biodiversity and ecosystem service outcomes at local levels. While considerable progress has been made in relevant fields (such as landscape ecology), our knowledge base required to make these assessments remains limited and deserving of increased interdisciplinary research effort.

Progress made towards the ultimate objectives of maintaining or increasing biodiversity values, ecosystem services, and human well-being at the landscape level will depend on management actions taken at more local scales, specifically at the individual farm, forest stand or small watershed level.

Conventional agricultural systems

Given the obvious importance of enhancing food security and sustaining agricultural livelihoods in most tropical landscapes, and the fact that forest conversion for agricultural uses remains a major threat to biodiversity conservation in most tropical countries, there is a clear need to identify and take advantage of biodiversity and ecosystem service enhancement opportunities in intensively managed agro-ecosystems (Swift et al 2001). Such opportunities may include improvements in conventional agricultural systems to enhance their sustainability and productivity by increasing the efficiency of water use, reducing dependence on chemical fertilizers and pesticides, and making better use of crop residues to enhance soil fertility. These and other sustainable agricultural practices can improve the biodiversity value of

¹ The stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfill, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems (definition adopted by the Ministerial Conference on the Protection of Forests in Europe (MCPFE) and adopted by the Food and Agriculture Organization of the United Nations (FAO).

croplands - particularly with relation to soil invertebrates, fungi and bacteria that are keys to soil formation and fertility - and thereby reduce pressures to convert forest lands or marginal habitats to agricultural production.

Agroforestry and silvo-pastoral systems, either those based on age-old practices found throughout Asia, Africa or Latin America, or more recent innovations oriented towards production of major agricultural commodities such as coffee and cacao, can contribute significantly to forest biodiversity conservation and provision of ecosystem services (Harvey et al 2008). Such systems, when compared to conventional field agriculture, can significantly improve soil fertility and local hydrological functions, reduce erosion, and natural pest management, and thereby contribute to provision of these key supporting, provisioning and regulating services (Daily 1997, Soto-Pinto et al 2002). The increased structural complexity of these systems which include forest trees as major components, provide important habitat for native forest flora wildlife, particularly in densely populated regions with highly fragmented forest cover (cf. Wunderle and Latta 1996, Daily et al 2003, Mayfield and Daily 2005, Harvey et al 2006, Sekercioglu et al 2007). Bhagwat et al (2008) present an interesting synthesis of 36 published studies in which the diversity of a variety of taxa (bats, other mammals, birds, reptiles, insects, trees, herbaceous and lower plants, and macrofungi in agroforestry systems and neighboring natural forest reserves throughout the tropics, were compared. Their findings indicated that the average species richness in agroforestry systems ranged from 62-139% (among the taxa groups compared) of that found in the adjacent natural forests, with similarity in species composition between agroforests and forest reserves averaging 25-65% among these taxa groups.

Traditional agro-forest management systems

Traditional agricultural systems, including shifting cultivation as well as traditional agroforestry systems such as the complex agroforests found in Southeast Asia and in many other tropical regions, are typically characterized by high diversity of cultivated crop species (and varieties) and local forest plant and animal species (c.f., Van Noordwijk et al 1997, Michon et al 2007, Ramakrishnan 2007, Rerkasem et al 2009). These management systems, based on traditional forest-related knowledge², are usually

² *Traditional forest-related knowledge*: “a cumulative body of knowledge, practice and belief, handed down through generations by cultural transmission and evolving by adaptive processes, about the relationship between living beings (including humans) with one another and with their forest environment”. Definition from the UNFF4 Report of the Secretary-General on Traditional forest-related knowledge (United Nations doc. E/CN.18/2004/7 (2004), adapted from Berkes et al. (2000)

tightly interwoven with traditional religious beliefs, customs, folklore, and community-level decision-making processes. They have sustained the cultures and livelihoods, and agricultural of local and indigenous communities throughout the world for centuries. They have also historically been dynamic, responding and adapting to changing environmental, social, economic and political conditions to ensure that forests and associated agricultural lands continue to provide tangible (foods, medicines, wood and other non-timber forest products, water and fertile soils) and intangible (spiritual, social and psychological health) benefits for present and future generations (c.f., Berkes et al 2000, Ramakrishnan et al 2002, ICSU 2002, Liang et al 2009).

Despite their importance and contributions to sustainable rural livelihoods, traditional forest-related knowledge and agro-forest landscape management practices are fast disappearing for a number of political, economic and social reasons. The negative implications of this loss of traditional forest-related knowledge on livelihoods, cultural and biological diversity, as well as on the capacity of forested landscapes to provide environmental goods and services, remain poorly understood, largely unappreciated, and undervalued by policy-makers and the general public in most countries (c.f., Michon et al 2007, Parrotta and Agnoletti 2007, Parrotta et al. 2009a, 2009b). Addressing the underlying causes of the erosion of traditional agricultural and forest management practices, and measures to expand their application, could yield important biodiversity benefits and enhance provision of forest environmental services in many regions (IAITPTF 2005).

Degraded and secondary forests

Perhaps the greatest opportunities for increasing biodiversity and increasing the capacity of forested tropical landscapes to provide environmental goods and services may be found in the extensive areas of secondary forests and degraded, formerly forested, lands that are becoming increasingly common worldwide (Lamb et al 2005). Since the beginning of the 20th century, an estimated 350 million hectares of tropical forests have been lost entirely, and another 500 million hectares of secondary and primary tropical forests have been degraded, i.e., altered beyond the normal effects of normal disturbance regimes, generally through unsustainable use (ITTO 2002). Within this category are: (1) degraded primary³

³ Primary forests are those that are considered to have never been subject to human disturbance, or much more commonly, have been used by indigenous and local communities with traditional lifestyles but have been so little affected by human-induced disturbance or extraction of forest resources that their structure, functions and dynamics have not undergone any changes that exceed the elastic capacity of the ecosystem.

forests whose structure and composition have been adversely affected by unsustainable harvesting of wood and/or non-wood forest products so that their species composition, ecological functions and dynamics are altered beyond the short-term resilience of the ecosystem; (2) secondary forests, or woody vegetation formations growing on lands that had been cleared of their original forest cover, and which typically develop through natural regeneration following abandonment after shifting cultivation, sedentary agriculture, pasture, or failed tree plantations; and (3) degraded forest land – former forest land severely damaged by the excessive harvesting of wood and/or non-wood forest products, poor management, repeated fire, grazing or other disturbances or land uses that damage soils and vegetation to a degree that inhibits or severely delays natural regeneration of forest after abandonment.

While they may not possess the same biodiversity value or yield the full array of forest environmental services as relatively undisturbed or sustainably managed primary forests, degraded primary forests and secondary forests are of considerable importance in most tropical landscapes. They play key roles in biodiversity conservation as well as provision of environmental goods and services to local communities. In many more densely populated regions, they are the only forests that exist. Long ignored or undervalued by ecologists and conservationists, but long appreciated by hundreds of millions of people in indigenous and local communities in Asia, Africa and Latin America who manage them as part of their shifting agricultural systems, the resilience and value of these forests for biodiversity and environmental services are increasingly recognized. Considerable attention has been given in recent decades to documenting, developing and sharing traditional and more recent management strategies for improving their contribution to sustainable livelihoods and biodiversity conservation (c.f., ITTO 2002, Lamb et al 2005).

The final category of formerly forested, but now degraded, lands presents both great challenges and great opportunities (Parrotta 2002, ITTO 2002, Lamb et al 2005). Historically, the most common response to forest land degradation has been abandonment or reliance on natural forest succession to restore lost soil fertility, species richness, and productivity. Periodic land abandonment has been the basis of sustainable traditional shifting cultivation and livestock husbandry worldwide. In many tropical regions, however, fallow periods are often shortened or eliminated due to increased population pressures and agricultural intensification. Without adequate inputs to recover lost soil fertility, productivity and land utility commonly decline. This has resulted in extensive areas of formerly forest lands in varying stages of degradation and fragmentation that require management to improve their capacity to provide environmental

services and contribute to biodiversity conservation, food and livelihood security.

Most tropical restoration efforts on degraded forest lands focus on the development of plantation and agroforestry, and conventional agricultural systems aimed at maximizing production of a very limited number of species of economic importance (particularly *Pinus*, *Eucalyptus* and *Acacia* in the case of plantation forestry). When successful, these efforts yield direct near-term livelihood benefits to landowners (or communities, for collectively owned lands) and can provide environmental goods and services such as wood for domestic energy or industrial pulpwood, soil stabilization, and carbon sequestration. However, neither agricultural development nor most past forms of reforestation have been sufficient to provide sustainable livelihoods and environmental services over the large areas of degraded land that have developed.

Some of these lands would naturally revert fairly quickly (years vs decades) to secondary forest if the pressures on them (i.e., biomass harvesting, grazing, fire, etc.) were lifted, particularly those that are located in proximity to biodiversity-rich native forests, and retain some residual trees, seedling banks, and soil seed stores composed of native species. There are well-documented examples where natural regeneration has occurred over very large areas in Puerto Rico (Aide et al 2000, Grau et al 2003), Costa Rica (Arroyo-Mora et al 2005), Brazil (Uhl et al 1988), Tanzania (Barrow et al 2002), and in many areas of South and Southeast Asia where effective community-based forest protection measures have been implemented (Poffenberger and McGeen 1996).

Other more severely degraded sites require some more active management, i.e., tree planting, to overcome the biophysical barriers that most commonly inhibit natural forest regeneration, such as topsoil loss, dominance by grasses and frequent fire (Parrotta et al 1997b, Parrotta 2002). Two broad strategies can be distinguished – those that focus primarily on biodiversity recovery (restoration plantings) and those that seek to provide goods and ecological services. The applicability of these strategies depends on the socioeconomic circumstances of the landholders (or stakeholders more broadly) as well as the ecological context.

Data from several studies have shown that restoration plantings can be quite effective for re-establishing functional, self-sustaining, biodiversity-rich forest ecosystems under appropriate circumstances (Parrotta et al 1997a, Tucker et al., 1997, Elliot et al 2003). One technique is to establish a small number of fast-growing but short-lived tree species to create a canopy cover that will shade out grasses or other weeds, reduce fire hazard, and facilitate, or catalyze, colonization (i.e., biodiversity enrichment) of the site by a wider range of species from nearby intact forests or forest fragments (Parrotta 1993, Elliot et al 2003). The success of this approach

in terms of biodiversity recovery depends on several factors. These include the degree to which the understory microclimate is altered within the planted forests (i.e., shading out of dominant grasses and other weeds, moderation of temperature and humidity fluctuations), improvements in soil fertility and organic matter development, and rates of post-planting colonization by native species from nearby intact forests principally through seed dispersal by frugivorous birds and mammals (Parrotta et al 1997b, Wunderle 1997).

An alternative approach involves planting of a much larger number of species representative of more mature forests, thereby short-circuiting often slow, uncertain natural succession processes. These plantings, more costly than those described above, are usually established at high densities (>2500 trees per ha). At these densities, the final forest composition is determined by competition among the planted trees and, under favorable circumstances, enrichment by colonization of additional species from nearby secondary or primary forests, or remnants. This approach has been used, very successfully, to restore native forests on bauxite minesites in the Brazilian Amazon, where over 160 species representing a range of life forms and successional stages were planted after mining and topsoil replacement (Parrotta and Knowles 1999, 2001).

Establishment of planted forests to provide a combination of income-generating goods (such as timber) and a larger range of environmental services than more conventional timber or pulpwood plantations is a second broad approach. Because of its likely financial benefits, it may permit larger areas to be reforested, although the biodiversity benefits may not be extensive (at the stand level) as more ambitious but less financially attractive restoration plantings. There is abundant evidence that tropical timber plantations (including those of exotic species), particularly those managed on long rotations, increase in biodiversity value over time through colonization by additional forest species (as described above) and creation of structurally diverse habitats for a wide variety of plant and animal taxa (cf. Parrotta 1997b, Carnus et al 2006, Brockerhoff et al 2008). There are many opportunities for improving the biodiversity value of planted forest landscapes established for income-generating purposes. They include greater use of native timber species, either in monocultures or mixtures, retaining or reestablishing native forest cover between plantation blocks, in riparian zones and elsewhere within large plantation landscapes, and uneven-aged management of more extensive plantations to increase the structural diversity among plantation units and create a more “natural” mosaic of planted and natural forest stands across the landscape (Lamb 1998, Lamb et al 2005, Brockerhoff et al 2008).

Concluding remarks - the importance of monitoring and adaptive management

Even when the considerable knowledge and experience of ecologists, forest and agricultural scientists and managers, and local people is used to develop landscape and site-level ecosystem restoration strategies, the outcomes will rarely be predictable. This unpredictability is due to the uncertainties inherent in dynamic ecological systems and the shifting priorities and needs of stakeholders. Given this degree of uncertainty, an adaptive management approach is called for, one that involves periodic monitoring of indicators of progress towards the attainment of environmental (i.e., biodiversity and ecological services) and socioeconomic goals. For restoration activities based on the reestablishment of forest cover, for example, the particular biophysical criteria and indicators adopted should be linked to site-specific objectives and goals. Indicators may include evaluations of tree survival, growth, and biomass productivity, population assessments of selected species or habitat diversity, understory crop yields, erosion rates, soil organic matter development or changes in soil physical and chemical properties, and frequency of fire or other major disturbances.

Similarly, socioeconomic indicators, such as incomes of farmers and other relevant stakeholders, access to adequate supplies of water for domestic and agricultural needs, biomass energy requirements, should also be included in such monitoring programs. These may be used to periodically evaluate the impacts of biodiversity/ecosystem services restoration activities to help guide changes in management practices to achieve optimal results and improve equity among stakeholders. The definition of appropriate criteria and indicators linked to specific environmental and socio-economic objectives, and a commitment to monitoring restoration outcomes to better inform future management practices through ongoing dialogue among stakeholders, is very useful. It can contribute greatly to managing the high degree of uncertainty in managing complex and dynamic tropical landscapes for the near- and long-term benefits to society and enhancement forest biodiversity and the ecosystem services they provide to people.

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Biodiversity of plantation forests and its relevance for ecosystem functioning

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Abstract

Plantation forests represent a small but growing proportion of the world's forests. This trend is likely to continue as climate change mitigation and 'carbon forestry' add to the motivation for afforestation. There are concerns that plantation forests negatively affect biodiversity because they are typically less diverse than natural forests and because some plantations replace natural forests, although land clearing for agriculture is the main driver of global forest loss. Effects of plantation forestry on biodiversity and methods for protecting biodiversity within plantations and surrounding landscapes have received much research attention in recent years. The conversion of natural forests and afforestation of natural non-forest land are mostly detrimental to biodiversity. However, in many countries afforestation occurs primarily on land previously cleared for agriculture, and this can assist biodiversity conservation by providing additional forest habitat, buffering edge effects, and increasing connectivity between forest patches. The use of native tree species is generally more beneficial for forest biodiversity, but even plantations of exotic trees can have a relatively rich understorey of native plants and associated fauna. However, exotic trees may spread beyond the planted area and can cause problems by becoming invasive species. Numerous opportunities exist for the protection and enhancement of biodiversity in the management of plantation forests, at both stand and landscape levels. Surveys for rare and threatened species and habitats of high conservation value should be conducted to ensure their specific protection. Forest biodiversity contributes to ecosystem functioning and the provision of ecosystem goods and services. For example, mixed forests are, on average, more resistant to outbreaks of forest pests than single-species forests. Furthermore, diverse forests are likely to be more adaptable to climate change. These and other relationships with biodiversity are explored, and opportunities for combining plantation forestry with biodiversity conservation objectives are highlighted.

Keywords: Plantation forest, biological diversity, mixed forests, ecosystem function, ecosystem services.

Introduction

Plantation forests and other planted forests represent a small but growing proportion of the world's forests. Plantation forests cover about 3.5% of the total world-wide forest area (ca. 140 million ha) and this is increasing by about 2–3 million ha per year, according to FAO's Global Forest Resources Assessment (FAO 2006a). This trend is likely to continue or even accelerate as climate change mitigation, 'carbon forestry' and the potential use of timber as a biofuel feedstock add to the motivation for afforestation, all of which may involve the establishment of plantation forests (Canadell and Raupach 2008, Lindenmayer 2009, Webster et al. 2010). Plantation forests are usually composed of one of a few tree species that are grown as pure, even-aged, intensively managed, fast-growing stands with clear-fell harvesting after relatively short rotations (typically ranging from 10–50 years). These forests are established either by seeding or planting of native or exotic trees commonly of broadleaved trees such as *Eucalyptus*, *Populus* (poplar), *Hevea* (rubber tree), and *Tectona* (teak), as well as conifers from the genera *Pinus* (pine), *Picea* (spruce), *Cryptomeria* (sugi or Japanese cedar), and various others (Carnus et al. 2006, FAO 2006a, Brockerhoff et al. 2008a). Plantation forests are most commonly grown to supply renewable raw material to wood-based industries but they also provide of a variety of other economic, environmental or social benefits either as their primary purpose or as a by-product. For example, afforestation can improve water quality and slope stabilisation in erosion-prone hill country, compared with agricultural land uses (Quinn et al. 1997, Knowles 2006). Hence, plantation forests can play a role in the provision of ecosystem goods and services that are important to society.

There are concerns that plantation forests may affect forest biodiversity negatively. This position may be related to the observation that plantation forests are often less diverse than natural forests (e.g., Barlow et al. 2007a, Pawson et al. 2008), although there are a number of factors that can confound such assessments (Brockerhoff et al. 2008a). Furthermore, some plantations replace natural forests (Cossalter and Pye-Smith 2003, Lindenmayer 2009), although land clearing for agriculture is generally considered to be the main driver of global forest loss. Over 90% of tropical forest loss that occurred in the 1990s was due to land clearing for agriculture and other uses (FAO 2001). For example, almost 10 million ha of oil palm plantations have been established in the last 20 years, mainly in Malaysia and Indonesia, and more than half of this expansion has occurred on land that was previously forested (Koh and Wilcove 2008, Wilcove and Koh 2010). Land conversion to plantation forest accounted for about 7% of tropical forest loss (FAO 2001). Deforestation and fragmentation of natural forests are among the main causes for the continuing decline of biodiversity (Brook et al. 2003, Lindenmayer and Franklin 2002, Laurance 2007). Deforestation is also a major contributor to climate change by causing as much as a fifth of total carbon emissions (UN-REDD 2008), which represents a further indirect threat to biodiversity. Ironically, some efforts to combat climate change by establishing planted forests to sequester carbon may lead to perverse outcomes when this involves clearing natural forests (Lindenmayer 2009). Therefore, it is critical to raise awareness that afforestation should not be undertaken at the expense of natural forests or other natural vegetation. Likewise, the protection and enhancement of biodiversity should be integral parts of plantation forest management. In many countries, plantation forests are already being managed responsibly, and mechanisms are in place to assist with this. Forest managers increasingly recognise the need to conserve biodiversity, and many adhere to sustainable management guidelines such as those of the Forest Stewardship Council (FSC) (Forest Stewardship Council 2010a, b), the PEFC, and voluntary guidelines (e.g., Shaw 1997, FAO 2006b). Biodiversity serves an important role in ecosystem functioning. Biodiversity can improve primary production, pollination, and resistance and resilience to disturbance such as that caused by insects and diseases, climate change, and other factors (McCann 2000, Loreau et al. 2001, 2002, Jactel and Brockerhoff 2007, Tylianakis et al. 2008). Given that plantation forests are often composed of a single tree species, the implications of relationships between biodiversity and ecosystem function are likely to be highly relevant for plantation forestry. For example, knowledge about effects of forest diversity can be applied to mitigate potential impacts of climate change or pests in plantation forests by creating plantations with species mixtures at the stand or landscape scale (Lindenmayer

and Franklin 2002, Brockerhoff et al. 2008a).

In the past decade, the International Union of Forest Research Organisations (IUFRO), the global network for forest science cooperation (www.iufro.org), has played an important role in the dialogue among scientists and policy makers regarding the role of forests in the conservation of biological diversity and the mitigation of land degradation and climate change. IUFRO and its members directly contributed as a connecting thread for enhancing forest-related synergies between the three Rio environmental conventions on climate change (UNFCCC), biological diversity (CBD) and land degradation (UNCCD). A major contribution was the IUFRO-led Global Forest Expert Panels (GEFP) initiative and assessment report on "Adaptation of Forests and People to Climate Change" (Seppälä et al. 2009). IUFRO input via the United Nations Forum on Forests (UNFF) facilitated cooperation with the CBD and the UNFCCC regarding forest-related linkages between forest biodiversity and climate change (e.g., for the CBD-UNFF Subregional Capacity-Building Workshop on Forest Biodiversity and Climate Change, Singapore, 2-5 September 2009). Priority areas for scientific cooperation that have been identified include, among others, the role of reforestation and afforestation in climate change mitigation and adaptation, local level climate-change impacts on forests, and monitoring and evaluation of adaptation measures (Seppälä et al. 2009).

With regard to the role of planted and plantation forests in sustainable forest management and, in particular, the conservation of biodiversity, there have been several IUFRO contributions. These included a report from members participating with the UNFF Intersessional Experts Meeting in Wellington, New Zealand, in March 2003, which was subsequently published in the *Journal of Forestry* (Carnus et al. 2006). Furthermore, this stimulated an IUFRO Working Party conference on "Biodiversity and Conservation Biology in Plantation Forests" in Bordeaux, France, in April 2005, a technical session on the same subject at the XXII IUFRO World Congress in Brisbane, Australia, in August 2005, and a contribution to the conference on "Ecosystem Goods and Services from Planted Forests" in Bilbao, Spain, in October 2006. Several key papers from these activities were published as a special issue of the journal *Biodiversity and Conservation* (volume 17, May 2008) including a synthesis paper (Brockerhoff et al. 2008a). For the purposes of the present paper we are drawing on these and numerous other resources.

The aims of this paper are, to:

- contribute to the FFPRI/OECD International Symposium for the Convention on Biological Diversity, regarding plantation forests in the context of forest biodiversity and ecosystem goods and services;
- provide background information for the upcoming Convention on Biological Diversity conferences CBD-SBSTTA-14 and COP 10;

- review the role of plantation forestry in biodiversity conservation;
- examine the potential role of plantation forests in reducing emissions from deforestation and degradation of forests;
- outline opportunities for the protection and enhancement of biodiversity in plantation forests to increase their value as habitat for biodiversity; and
- assess linkages between forest biodiversity and ecosystem functioning, specifically with regard to resistance to disturbance factors such as invasive species and climate change.

Positive and negative effects of plantation forests on biodiversity

Much research attention has been given in recent years to the effects of plantation forestry on biodiversity within plantation forests and the surrounding landscapes. Recent reviews of this topic include Hartley (2002), Lindenmayer and Hobbs (2004), Carnus et al. (2006), and Brockerhoff et al. (2008a). Although the use of native tree species is generally more beneficial for forest biodiversity, even plantations of exotic trees can have a relatively rich understorey of native plants (Brockerhoff et al. 2003, Eycott et al. 2006) and associated fauna, including species of conservation concern (Brockerhoff et al. 2005, Pawson et al. 2010). This role of plantation forests is particularly important in regions that have suffered substantial forest loss, where plantation forests may become critical refugia for threatened species (Brockerhoff et al. 2005). However, exotic trees may spread beyond the planted area and can cause problems by becoming invasive species (Richardson and Rejmanek 2004), with flow-on effects on biodiversity in affected habitats (Samways et al. 1996, Pawson et al. submitted).

There is good evidence that the conversion of natural forests and afforestation of natural grasslands and other natural non-forest land can be detrimental to biodiversity (Barlow 2007a, Carnus et al. 2006, Brockerhoff et al. 2008a, Gardiner et al. 2008) and alter ecosystem processes (Barlow et al. 2007b). However, in many countries afforestation now occurs primarily on land previously cleared for agriculture (e.g., Brockerhoff et al. 2008b). Under those circumstances, plantation forests may integrate well into the landscape matrix of natural forest. Here, afforestation of agricultural land can assist biodiversity conservation by (i) providing surrogate habitat for forest species, (ii) buffering against edge effects between forest remnants and cleared forest land, and (iii) increasing connectivity between forest patches.

Plantation forests as surrogate forest habitat

Plantation forests can provide surrogate forest habitat for a proportion of forest generalist and

forest-dependent species (Mesibov 2005, Brockerhoff et al. 2008a, Pawson et al. 2008, Felton 2010, Paquette and Messier 2010). The value of plantations as alternative habitat for native species is dependent on the ecological traits of individual species. Plantations tend to be more important for shade-tolerant plant species that can regenerate within the time frame of a single plantation rotation (Brockerhoff et al. 2003); however, clearfells and young stands are important for early-successional species and those associated with more open habitats (Eycott et al. 2006). In New Zealand, plantations can support high densities of insectivorous birds, although fruit and nectar feeders are scarce given the lack of suitable food resources in a Pinus-dominated canopy (Clout and Gaze 1984, Deconchat et al. 2009). Invertebrates tend to be dominated by generalist species (predators, detritivores, and fungivores) whereas many monophagous or oligophagous herbivores and their associates may be absent due to a lack of suitable host plants. Obviously, the impact of these ecological traits on colonisation of plantations will be dependent on several biogeographic and other context-dependent factors (Brockerhoff et al. 2008a), including the original land cover type, proximity to remnants of natural vegetation, the identity of the planted tree species, and how the forest is managed.

Plantation forests as buffers against edge effects

Forest fragmentation is often detrimental to biodiversity because it reduces the amount of available habitat (Watson et al. 2004; Benedick et al. 2006) and causes an increase in edge habitat which creates unfavourable micro-climatic conditions and opportunities for biological invasions from adjacent non-forest habitat (edge effects) (Yates et al. 2004, Denyer et al. 2006). Plantation forests adjacent to natural forests can be beneficial by buffering abiotic and biotic edge effects (Denyer et al. 2006). Conceptually plantations could protect remnant native ecosystems from invasive exotic species that are present in neighbouring productive land uses, such as agricultural systems, by blocking the environmental cues that mediate dispersal processes. Such spillover of agriculturally subsidised predators is known to occur (Rand et al. 2006), however, as yet the ability of plantations to protect remnant native ecosystems from such effects has not been thoroughly tested. Conversely, plantation forests may also act as a source of biological invaders. This may concern invasive species living within the plantation or even the planted tree species itself (see below).

Plantation forests and landscape connectivity

Forest fragmentation generally increases the isolation of forest patches (van der Ree et al. 2004), which may reduce gene flow between meta-populations of species, particularly those that are poor dispersers and lack the ability to move across non-forest land. Here, plantation forests may assist by providing forest-type

conditions between forest remnants, thereby increasing connectivity between forest patches (Hewitt and Kellman 2002a, Creegan and Osborne 2005, Tomasevic and Estades 2008). This may be particularly important in landscapes where forest loss has reached critical levels and may result in extinction of forest species that are sensitive to habitat fragmentation. Species differ in their sensitivity to forest loss, and some bird species may already be threatened by local extinction when habitat loss exceeds approximately 65% (i.e., 35% remaining forest cover) (Villard et al. 1999). Zurita and Bellocq (2010) found that native forest habitat loss was the major determinant of changes in forest bird communities in the Atlantic forests of Argentina and Paraguay. The effects of forest loss on community composition were not as pronounced in landscapes where the matrix habitat was predominantly plantation forest as opposed to agriculture (Zurita and Bellocq 2010). However, such a beneficial effect was not as clearly apparent in another study assessing the effects of different forest and non-forest habitats surrounding native forest remnants and regenerating forest patches (Deconchat et al. 2009). In landscapes dominated by grassland, large-scale afforestation may have the opposite effect, reducing connectivity for open-habitat species.

Role of plantation forests in reducing emissions from deforestation and degradation

Intensively managed plantation forests are highly productive and play an important role in the global supply of forest products. Although plantation forests currently represent less than 4% of the world-wide forest cover, they provided about 35% of the global roundwood supply in 2000, according to the Millennium Ecosystem Assessment (MEA 2005a). It is expected that their importance will increase further and that by 2020, about 44 % of global roundwood will be supplied by planted forests (MEA 2005a). Eventually they may meet much of the future global demand for wood (Paquette and Messier 2010). With their more efficient production of timber and other forest products, requiring comparatively little land, it is conceivable that plantation forests can make a considerable contribution to the protection of natural forests, by reducing the need for natural forests to provide these products. Several authors have reviewed this topic, including Sedjo and Botkin (1997), Hartley (2002), Carnus et al. (2006), Brockerhoff et al. (2008a) and references therein.

There are examples of countries where plantations have relieved pressure on natural forests. For example, in New Zealand well over 90% of the domestic timber consumption is produced from plantation forests while natural forests, which still cover nearly a quarter of the land area, are now largely protected (Brockerhoff et al. 2008a). Furthermore, because New Zealand is a major net exporter of roundwood and other forest

products, it has been argued that New Zealand's plantation forests also reduce the pressure on harvesting natural forests elsewhere (Maclaren 1996), although there also appears to be a noticeable increase in imports of tropical timber (and furniture manufactured from such). However, in other countries an expansion of plantation forestry has not led to a reduction in extraction from natural forests and/or deforestation (e.g., Clapp 2001, see also Paquette and Messier 2010).

Obviously, the establishment of plantation forests does not necessarily result in the concomitant protection of natural forests (Hartley 2002), unless this is achieved by way of a managed, co-ordinated process that integrates land use at a regional (or national) scale (Paquette and Messier 2010). Similarly, the expansion of plantation forestry does not necessarily influence other drivers of deforestation such as agricultural development. For example, the large-scale development of oil palm plantations in Malaysia and Indonesia, about half of which are considered to replace previously forested areas (old growth and other forests) (Koh and Wilcove 2008, Wilcove and Koh 2010), is probably independent of the development of plantation forests elsewhere in these countries. However, the increasing demand for biofuel feedstocks from plantations and the planting of new forests to sequester carbon for climate change mitigation have the potential to cause further deforestation unless these new drivers of land use change are well managed (Lindenmayer 2009). The United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries ('REDD') (UN-REDD 2008) aims at increasing the profitability of protecting forests from clearance and exploitation. Plantation forests may have a role in this by providing an alternative source of forest products (as outlined above), but ensuring that plantations really assist with the protection of natural forests and contribute to mitigating climate change (rather than providing a cause for emissions from replacing natural forests) is a complex task (Paquette and Messier 2010).

Options for protecting and enhancing biodiversity in plantation forests

Numerous opportunities exist for the protection and enhancement of biodiversity in plantation forests (Hartley 2002). Enhancement strategies can be applied at the stand or landscape level, reflecting the geographic scale at which they operate (Lindenmayer and Hobbs 2004, Brockerhoff et al. 2008a). Among possible actions operating at the stand scale, the selection of tree species can have a significant effect on the value of plantation forests as habitat for biodiversity. Careful selection of species and the use of mixed species planting can provide even greater opportunities for biodiversity by providing resources such as nectar and fruit and structures important as

micro-habitat (Hartley 2002, Brockerhoff et al. 2008a). Silvicultural management often prevents the development of a multi-tier canopy (Carnus et al. 2006) and can reduce or eliminate the presence of over-mature, or recently dead trees (snags), which is detrimental to biodiversity. Increasing stand structural complexity can improve the abundance and species richness of birds in plantations (Najera and Simonetti 2010). 'Biological legacies' such as snags can be created during harvesting (e.g., by leaving some stumps cut at a height of 5 m) to provide conditions during the subsequent rotation that are suitable for cavity-nesting birds as well as important habitat for saproxylic beetles and fungi (Lindhe et al. 2004, Lindhe and Lindelöw 2004). Stand age is also a significant predictor of biodiversity value as older stands frequently support more native forest species (Brockerhoff et al. 2003, Lindenmayer and Hobbs 2004, Taboada et al. 2008, Pawson et al. 2009), although this is context dependent. If plantations were established on former open habitats, such as heathlands or grasslands, then young stands and clearfells may be more important for the original flora (Eycott et al. 2006) and fauna. Rotation length (i.e., the time between planting and harvesting) is largely influenced by species- and site-specific characteristics (e.g., tree growth) and economic factors.

It is possible to cater for both open-habitat and forest specialists by managing forests such that they always include a range of young and older stands as well as open areas, and by protecting remnants of wetlands, natural grasslands and native forest that often occur within plantation forests. Historically, attempts to enhance biodiversity in plantations have primarily focussed at the stand level. However, substantial changes have occurred in the way we perceive fragmented landscapes and, in particular, the role of 'matrix' habitat which separates remnant native ecosystems. A reinvestigation of the processes by which landscape-scale habitat complexity affects biodiversity indicates that plantation forests are no longer necessarily seen as inhospitable habitat. Conversely, plantations may represent a relatively 'soft' matrix that can support native species and facilitate dispersal between native habitat remnants (Fischer and Lindenmayer 2006, Brockerhoff et al. 2008a). The complexity of this matrix can be enhanced by the juxtaposition of different plantations types (e.g., tree species), stand ages, stand size and shape, and silvicultural management such as pruning or thinning. Furthermore, the proximity and size of remaining native forest fragments in the landscape can be an important predictor of biodiversity within plantation stands (Hewitt and Kellman 2002b, Pawson et al. 2008). Plantation forests often include remnants of natural forests and other vegetation types, and the protection and restoration of these habitats is of great importance.

In addition to general considerations regarding stand and landscape level factors, there are various special

cases including the occurrence of rare and threatened species and habitats of high conservation value. Such cases are often context-specific and require adaptive approaches to management. The presence of rare and threatened species in plantation forests is not as uncommon as one might expect. For example, a recent survey in New Zealand recorded 118 species classified by the Department of Conservation as threatened that are associated with plantations forests (Pawson et al. 2010). Surveys should be conducted to identify threatened species in plantations to enable appropriate management actions designed to ensure their persistence.

Biodiversity and ecosystem function: Provision of regulating services

It is widely acknowledged that biodiversity can play a critical role in driving forest ecosystem functioning (Scherer-Lorenzen et al. 2005). The Millennium Ecosystem Assessment (MEA 2005b) listed 25 different provisioning, cultural, supporting, and regulating ecosystem services that may be affected by biodiversity changes. As a consequence, there is considerable concern that tree monocultures would be less sustainable than mixed forests (Kelty 2006, Thompson et al. 2009). Indeed, an increasing body of evidence supports the view that there is a positive relationship between tree species diversity and forest productivity. In a recent review, Thompson et al. (2009) found that in 76% of the 21 studies that addressed the issue, stand production was higher in mixed than in pure stands, including in forest plantations (e.g., Parrotta 1999, Piotto 2008 in Thompson et al. 2009, but see also Potvin and Gotteli 2008). However some other studies found no significant effect of tree diversity on wood biomass production which confirms that the identity and the relative abundance of tree species in mixtures may be more important than the number of tree species per se. Mixed forests may also be more productive because of a reduced impact from pests and pathogens. In a recent meta-analysis of the international literature, Jactel & Brockerhoff (2007) found that overall, mixed stands are less prone to insect herbivory. Other studies indicate that this is also the case for fungal pathogens and mammalian herbivores (Jactel et al. 2008). The positive relationship between tree species diversity and forest resistance was more evident for insect specialists (monophagous species) than for generalists (polyphagous species). Again the composition of the mixture was relevant since the association of tree species with more contrasting functional traits resulted in higher level of resistance, notably because most insect herbivores lose some fitness when they shift from their 'natural' host trees to other, phylogenetically distant tree species (Bertheau et al. 2010). Remnants of old forests may also contribute to the regulation of forest pests by providing critical resources such as cavities which some birds require as

nest sites (e.g., Barbaro et al. 2008). Such cavities do typically not occur in plantation forests managed in short or medium-length rotations. Therefore, cavity-nesting birds may be more abundant in plantation landscapes that include such forest remnants, allowing them to live and forage in such landscapes. The association of several tree species and the corresponding partial functional redundancy may also result in higher resilience to disturbance, according to the insurance hypothesis (Yachi and Loreau 1999). Fridley et al. (2007) found a negative relationship between local (e.g., within-stand) diversity of native plant species and invasion by exotics, which may be explained by the higher number of vacant ecological niches that can be occupied by invasive species in more simple ecosystems.

The frequency of natural disturbances (storms, droughts...), pest outbreaks and species invasions is likely to increase as a consequence of global change. Diverse forests are likely to be less affected by climate change because in a mixed forest, some tree species are probably able to adapt to changed climatic conditions and also because of facilitation processes. There are opportunities for combining plantation forestry (and 'carbon forestry') with biodiversity conservation objectives, allowing for co-benefits between production, conservation, and risk aversion. Increasing tree species diversity may well represent one of the most promising ways to improve the sustainability of plantation forestry in the face of these threats.

Conclusions and outlook

Plantation forests are becoming an increasingly widespread land use in many parts of the world. As pressure on natural forests is likely to continue, the role of plantation forests in meeting a wide range of ecosystem goods and services will become more important, well beyond the provision of timber and other forest products. Plantation forests will have a more prominent role in the conservation of biodiversity, and our ability to create and manage biologically more diverse plantations will be critical in ensuring the adequate functioning of these novel forest ecosystems. Large-scale plantations of single tree species are likely to be less adaptable to the changing environmental conditions of the 21st century which will present challenges including the potential intensification of biological invasions, outbreaks of pests and diseases, and a changing climate. We are well advised to anticipate and respond proactively to these challenges.

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Tropical forest management and sustainability issues

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Abstract

Continuing to search for a globally accepted definition of sustainable forest management seems pointless. Even if we could agree on what we mean by “sustainable” applying the concept, and achieving the desired outcomes face many problems. Trying to satisfy multiple and often disparate objectives, each with differing timeframes and spatial extents, is one complication. Attempting to accommodate varying environmental, economical, social, and political conditions, many of them outside the reach of forest management, is another. Rather than aiming for an unattainable and contentious ideal, it may be more useful to strive for continuous improvement to achieve better outcomes when the best is unachievable. Such an approach would tailor both research and management to the relevant features of the environment and background conditions. Research could also be scaled more appropriately, taking into account more realistic local ecological and management timeframes and spatial extents. By looking for ongoing improvement in management, rather than for some distant and probably unattainable ideal or targets, planners, managers, and researchers may be able to ensure a more sustainable use of forest resources.

Keywords: tropical forests, biodiversity, resilience, targets

Tropical forests, biodiversity and resilience

For the Convention on Biological Diversity (CBD) the term biodiversity “means 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.” From this it is clear that forest biodiversity is integral to sustainable ecosystem functions and, therefore, vital for the availability of goods and services, from tourism to timber and non-timber products. It is important to consider carefully this dichotomy between biodiversity and diversity. Many see these concepts as roughly synonymous and assume that high local species richness is desirable in itself (see Sheil et al., 1999 or Thompson et al., 2009). Hence, “species richness,” or “processes that maintain richness” are assumed to be management priorities (e.g., Stork et al., 1997). Some scientists however will suggest that all species should

be counted equally, or weighted on their taxonomic “distinctiveness,” or vulnerability, or some such combination; whereas others might choose to weigh those with potential commercial values more highly, or according to their perceived public appeal. Yet others might take an ethics-oriented stance, placing intelligent species higher. Others may favour species named in religious texts, or those that serve as national or state symbols, represent local clan totems, or hold personal relevance. There is no “right” answer and alternatives cannot be resolved by appeals to science. They will be best addressed through informed consensus (Sheil et al., 2004).

Forest biodiversity provides a vast array of goods and services: timber, non-timber forest products, tourism and recreation opportunities, a storehouse for genetic resources, an insurance against extreme events, etc. Conservation of forest biodiversity appears therefore as a pre-requisite for the conservation of the complete array of forest ecosystem functions. A recent review by the CBD (Thompson et al., 2009) concluded that maintaining and restoring biodiversity in forests promotes their resilience to human-induced pressures and is therefore an essential “insurance policy” and safeguard against expected climate change impacts. Increasing the biodiversity in planted and semi-natural forests will have a positive effect on their resilience capacity and often on their productivity (including carbon storage).

Yet, conserving this complete array may not be necessary or realistic. Forests are diverse and made of diverse interacting species that provide a wide range of goods and services, but from a human viewpoint, a very high diversity is not necessarily linked to more useful or valuable forest goods and services. Less species does not mean less important as shown by monodominant *Caesalpinaceae* forests in Central Africa. These are not less valuable than the incredibly diverse lowland rainforests of Papua New Guinea. On the contrary, the “oligarchic” forests dominated by few species of abundant fruit bearing trees (e.g. *Myrciaria dubia*) or palms (e.g. *Euterpe oleracea*) have often a higher economic value than more diverse forests; the same is also true for timber or wood products (e.g. tropical vs. boreal forests) (Nasi et al., 2002). Recent reviews have also shown that secondary forests can play an important role in biodiversity conservation in the tropics particularly when intact or old-growth forests are nearby though because abundance, geographic range and levels of habitat specialization are often related, habitat generalists

might dominate even if relatively rare old-growth specialists are present (Dent and Wright, 2009).

Nevertheless, functional diversity is crucial as shown by a review of the evidence of regime shifts in terrestrial and aquatic environments in relation to resilience of complex adaptive ecosystems and the functional roles of biological diversity. This review showed the likelihood of regime shifts increase when resilience is reduced by such actions as removing response diversity capacity, removing whole functional groups of species, or altering the magnitude, frequency, and duration of disturbance regimes. The combined and often synergistic effects of those pressures can make ecosystems more vulnerable to changes that previously could be absorbed. As a consequence, ecosystems may suddenly shift from desired to less desired states in their capacity to generate ecosystem services because of a reduced functional diversity (Folke et al., 2004).

The traits that have been important for species survival or that make some species more vulnerable to extinction appear consistent across time scales. Terrestrial organisms (e.g. tropical trees or animals) are more extinction prone than marine organisms; mutualist species, trophic specialists and phylogenetically specialized groups may be collectively more prone to extinction than generalists; plants that persist through dramatic changes often reproduce vegetatively and possess various mechanisms of die back (Stork et al., 2009).

How well protected and/or managed are tropical forests nowadays?

Protected areas are the cornerstones of most national and international conservation strategies, providing refuges for species that cannot survive and ecological processes that cannot be maintained in intensely managed landscapes or seascapes (Dudley et al., 2005). The statutorily protected area (International Union for Conservation of Nature categories I-IV) in tropical and subtropical realms, estimated at about 1.9 million km² (Schmitt et al., 2009), has more than doubled over the past decade reaching or exceeding the CBD "10%" target for several biomes (Coad et al., 2009). Even with such an increase, the effectiveness of these areas alone in protecting biodiversity is somewhat limited. For example if Brazil's Amazon Region Protected Areas project is fully implemented, the area receiving statutory protection will increase to nearly 2.36 million km², or 46% of the Brazilian Amazon (Azevedo-Ramos et al., 2006), but will only encompass about 30% of the ranges of mammal species. Despite almost ideal conditions for representing biodiversity within the protected areas network in China, surveys show that the majority of giant pandas (*Ailuropoda melanoleuca*) continue to live outside protected areas (Louks et al., 2001).

The problem is global as the recent growth of the global protected-area network has not been planned or

managed to maximize the overall conservation of biodiversity (Rodrigues et al., 2004). Many areas with outstandingly high conservation value are either unprotected or receive limited protection and under serious threat; yet prospects of significantly expanding the network are not really bright for various reasons, one of which is the people vs. park debate (Galvin et al., 2008).

It seems more realistic to work with current land users that have managed this land for a long time (Robichaud et al., 2009), at least in some areas, than to expect many new reserves to be set up. New reserves would likely cause some displacement of local people, or curbs on their activities, thereby creating more people-conservation conflicts. Strict preservation, in which local people are excluded or prevented from harvesting forest products, not only fosters resentment against conservation, but it also diminishes indigenous knowledge that could contribute to ecosystem management (Laumonier et al., 2008). For tropical developing countries, rather than being induced or pressured into abandoning their forest heritage, rural communities should be actively involved in managing forests productively (Wood, 1995; Michon et al., 2007) using multipurpose management schemes. As local perceptions do not always accord with conventional scientific understanding, this is something that requires discussion and trade off (López-Hoffman et al., 2006).

Protected areas will also only function effectively as tools for conservation if properly managed to retain their constituent species and habitats. Unfortunately, several surveys (Stolton and Dudley, 1999; Carey et al., 2000; Bruner et al., 2001; Dudley et al., 2004) found widespread threats, particularly in the tropical countries including particularly illegal use related to logging and poaching. To protect each area of the global network more effectively would indeed entail considerable direct and opportunity costs to most developing countries (Wilkie et al., 2001; Blom, 2004).

Having 15% of the world's land surface effectively protected means however that 85% remain outside of any formal protection. Just as the area of protected tropical forests has increased, so too has the area of tropical forests under formal management, from less than one million ha in 1988 (Poore et al., 1989) to about 36 million ha in 2005 (ITTO, 2006). Tropical forests and plantations certified under the Forestry Stewardship Council (FSC) standards have increased from less than 0.5 million ha in 1998 to 17.9 million ha in 2009 or 15.3% of the total area certified by the FSC globally (FSC, 2009). Within the tropics, almost 82% of the certified area is natural forest; the balance consists of plantations (Nasi and Frost, 2009).

Yet despite considerable investments and progress in tropical forest management, the results in terms of changed silvicultural and land-use practices have been modest (Wunder, 2006 and references therein). The original tenets of forest management introduced to the

tropics in the 1950s and based largely on European experiences, have not really changed with the times and different circumstances in which forests are now managed. Sustained yield harvesting remains the norm, maximizing yield within the imposed constraints of a minimum cutting diameter for the harvested species and a minimum rotation period among the harvested sections of forest, all across multiple cutting cycles (Nasi and Frost, 2009).

Management paradigms: is sustainability achievable?

The principles underpinning forestry management have shifted over time. The sustained yield forestry concept (SYF), achieving sustained production of a single commodity, almost always timber, served its purpose for a long time but proved inadequate both conceptually and practically in satisfying societal demands on forests (Nasi and Frost, 2009). New societal demands on forests derived from the sustainable development discourse pushed the SYF paradigm further aside and led to one based on the sustained production of multiple goods and services (multiple use forestry, MUF). This was soon extended to include provisions for maintaining future options and not damaging other ecosystems (sustainable forest management, SFM) as forests produce much more than just timber, and the interests at stake extend beyond those of logging companies, timber merchants, silvicultural managers or researchers (Michon et al., 2007). This was further supported by the rise of certification, designed to assure consumers that the products being purchased were being produced sustainably, equitably, and with appropriate management (Upton and Bass, 1995). The limited success of MUF and SFM (Gong, 2002; Hammond and Zagt, 2006), seen against the backdrop of ongoing tropical deforestation, led in turn to the need to look beyond the forests and to consider the integrated management of land, water and living resources. This approach, sometimes called integrated natural resource management (INRM), was designed to achieve both conservation and sustainable use in an equitable way at the landscape level and is somewhat the application to forests of the CBD Ecosystem Approach.

At each stage unfortunately, management became inherently more complicated, necessitating to recognize complex dynamics, with their attendant shifts and thresholds, inherent uncertainties, and the combined influence of both social and ecological forces on the outcomes. At the same time, one cannot help notice that forest management methods and prescriptions have only evolved marginally from the beginning of industrialization. Even with progressive approaches such as MUF or SFM, the multidimensional aspect of tropical forest management is still too often defined by specialists from other regions and cultures (Michon et al., 2007).

Whether any such management is really sustainable is yet to be determined, given the long timeframes involved in natural forest dynamics. Current levels of extraction of the main timber species in the Congo Basin are probably not economically sustainable, as the volumes of commercially important species are unlikely to recover within a cutting cycle. The volumes extracted would need to be reduced by between a quarter and a half to be sustainable for the next rotation but then it might become uneconomic (Karsenty and Gourlet-Fleury, 2006). Similar concerns about the harvesting intensity needed to obtain a sustainable yield, and the often lower extraction level needed to be ecologically sustainable as well, have been expressed about other tropical forest ecosystems, e.g. mangroves (López-Hoffman et al., 2006), *Tabebuia* spp. (Schulze et al., 2008). Interestingly the converse also applies. Current prescribed logging intensities may be too low to create the conditions for the regeneration of many important commercial species that currently support much of the timber industry. This creates a paradox, in which the intensity of logging of these species is too high, but the overall logging intensity is too low to create the kind of disturbance regime needed for the regeneration of these preferred species (Fredericksen and Putz, 2003; Karsenty and Gourlet-Fleury, 2006).

Large areas of tropical forest, irrespective of whether protected areas or controlled by communities, are still being harvested unsustainably, if not illegally, through logging, hunting, or collecting non-timber forest products. Part of the reason is that the legal frameworks are often antiquated, inadequate, underfunded, and poorly enforced. Corruption is generally widespread at all levels in natural resource management (Kolstad and Søreide, 2009). For example, flaws in the legal framework in Cameroon allow logging companies to ignore some of the most harvested species in their forest management plans (Cerutti et al., 2008). As a result, much of the annual timber production is being realized as if no management rules apply. Because the government lacks the capacity to draw up management plans, something for which it is legally responsible, this task has been delegated to logging companies. Not surprisingly, under this arrangement, silvicultural elements and economic concerns take precedence over environmental and social ones.

In addition, many tropical forests, especially in Latin America, face increasing pressures of spontaneous colonization and conversion of forest to agricultural land (Geist et al., 2006). People in and around recently established reserves can face constraints on their use of resources and are contesting this, as are those who were displaced from their ancestral lands in the past (West et al., 2006; Adams and Hutton, 2007; Agrawal and Redford, 2009, and references therein). Some displacement does continue, though the exact scale, cause and consequences are fiercely debated (e.g. see Schmidt-Soltau, 2009 and

Curran et al., 2009). Some actors are using this discontent as camouflage for their own larger ambitions. In short, managing tropical forest reserves can be riven with conflict, disparate objectives, and lack of agreement on how best to move forward.

Some lessons learned

We must learn to adapt our management to the emerging new modified ecosystems we created and not only focus on so-called “natural” or near pristine systems. We should envision SFM as a co-evolutionary process between the changing forest, the changing market and an industry moving towards higher efficiency standards over time. Our aim should be to maintain functional forest ecosystems that provide a continuous flow of goods and services for the benefit of everybody, especially the poor and marginalized people.

Broader management models are needed for tropical production forests. We need new, innovative models of tropical forest management, based on locally-appropriate paradigms and application, in which the concept of sustainability is set in the broader context of societal demand on tropical production forests. A viable network of protected areas is necessary for conserving some tropical forests and their biodiversity but it is surely insufficient. Because of costs (direct and opportunity), practicality and the pressures of competing interests and land uses mean that the network can never be extensive enough to encompass all the biodiversity that needs protecting. The battle to conserve tropical forest biodiversity will therefore be won or lost in managed forests being used to produce timber, non-timber forest products and services. Selectively logged forest, managed appropriately, can provide habitat for otherwise threatened species and complement protected areas (Meijaard et al., 2006; Clark et al., 2009).

Production forests therefore need to be managed for timber production, but also for objectives such as supporting local livelihoods, biodiversity conservation, and environmental services, including carbon capture and storage (ITTO, 2002; Nasi and Frost, 2009; ITTO/IUCN, 2009). Management perspectives will need to embrace the larger landscape and not be focused simply at the stand level (Frost et al., 2006). For conserving biodiversity, this requires thinking in terms of managing the landscape as a continuum of patches, corridors, and matrices, at a range of scales, rather than as a strongly differentiated patchwork at one scale (Fischer and Lindenmayer, 2006). For livelihoods, it means looking beyond agricultural land to forests as: i) mosaic of intact forest patches, from which people obtain various goods and services, including sustenance in times of hardship; sacred groves, which remain untouched; ii) ancient agroforests; iii) land that may yet be cleared for settlement and cultivation; and iv) patches

regenerating after abandonment. Managing such complexity for both livelihoods and biodiversity conservation is still in its infancy (Frost et al., 2006; Pfund et al., 2008).

Avoiding irreversible change is more relevant than striving for sustainability

Trying to achieve “sustainability” appears a noble but misplaced and ultimately unrealistic goal. No matter what actions are taken forest composition and structure inevitably change with time, both in response to endogenous processes (e.g. forest succession) and external pressures (e.g. changes in rainfall and temperature regimes, human disturbances). Species respond to environmental change individually rather than synchronously as communities or ecosystems (see Davis and Shaw, 2001 or Engelbrecht et al., 2007). Some species are lost or become rare while new ecosystems emerge with new combinations of species, interactions, and properties. Relieving modern stressors, such as logging or hunting, will not necessarily result in these altered ecosystems revert to their original state (O'Neill, 1998; Hayashida, 2005).

We should aim avoiding irreversible change, especially deliberate or inadvertent conversion to non-forested land. Given that some change is inevitable, the aim should be to manage for resilience--the capacity of forest composition to change without any radical shift in overall structure and function.

Uncertainty exists in all of this, especially in those systems or parts of systems that are driven by external forces of climate and human demand. Because of this, management needs to be flexible, taking into account new knowledge and understanding, changing circumstances, and based on learning lessons from present practices, both locally and elsewhere. The inextricable link between people and the environment must also be recognized and taken into account. No doubt, this all adds to the complexity of management, but decisions on action cannot be deferred. They have to be taken on best available information, with careful assessment of the potential costs and risks, and a commitment to monitoring and assessing outcomes, and learning and applying the lessons, where possible.

Unrealistic or ill-defined targets are of little use.

Targets are important to provide well defined outcomes to be achieved in a given amount of time. However unrealistic targets are of little use. In 1990, International Tropical Timber Organization (ITTO) members agreed to strive for an international trade of tropical timber from sustainably managed forests by 2000: a commitment known as the “Year 2000 Objective”. A significant part of the ITTO program of projects and activities was devoted to its achievement. The assessment made in 2000 showed that tropical countries had made significant progress in the formulation and adoption of policies but less evidence

was found of progress in implementation.

Recognizing this lack of progress, ITTO members re-stated their commitment to moving as rapidly as possible towards achieving exports of tropical timber and timber products from sustainably managed sources, renaming this commitment as "ITTO Objective 2000". It remained a central goal of the Organization, supported by renewed efforts to raise the capacity of government, industry and communities to manage their forests and add value to their forest products, and to maintain and increase the transparency of the trade and access to international markets.

In 2005, the assessment (ITTO, 2006) showed progress with a minimum of 36.4 million ha (4.5%) of the total natural permanent forest estate (814 million ha) are considered to be under SFM with an estimated 96.2 million ha (27%) of natural production forests are covered by management plans. We must applaud the progress but also recognize that we are still very far from the target and far beyond the deadline. Given the very ambitious targets proposed for discussion for the CBD (SCBD, 2010) – e.g. Target 3: By 2020, subsidies harmful to biodiversity are eliminated and positive incentives for the conservation and sustainable use of biodiversity are developed and applied; Target 5: By 2020, the loss and degradation of forests and other natural habitats is halved; Target 7: By 2020, all areas under agriculture, aquaculture and forestry are managed sustainably; Target 12: The extinction of known threatened species has been prevented – one needs to be prepared to monitor progress and accept that targets might not be achieved in 2020 under the business-as-usual scenario or to propose ways and incentive to change this. Communication has to be very clear and proactive so that we do not open ways to some constituencies to criticize indiscriminately and to use alleged non-achievement as an excuse to throw the baby with the bath water.

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Sustainable Forest Management (SFM) and biological diversity under changing needs of society - an example from the European situation

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Abstract

The demand for new forest products, forest services and forest information by various sectors of society continuously increases. To date, especially the energy sector with the increasing use of wood-based bioenergy, the construction sector using wood as an environmental sound material, and the health sector using medicinal compounds from forests for therapeutic means, all require new products, services, and up-to-date forest information. In addition there is also an increasing need to serve societies and decision making bodies with forest services and forest information for cross-sectoral thematic areas such as: climate change, conservation, forests and water, green public procurement policy, and various environmental ecosystem services. This article provides a proposal on how the present MCPFE¹ Criteria and Indicator set on Sustainable Forest Management (SFM) could be adjusted to serve better society's needs by combining information of the forest and other sectors and related thematic areas. The MCPFE forest indicators have been used to evaluate the status of Europe's forest biodiversity. This proposal for a wider use of the MCPFE Criteria and Indicators covers the following items: updating quantitative and qualitative indicators in accordance to new demands, review of overall policies and special policy areas, setting of threshold values, and verification issues. Furthermore this article discusses the key forest biodiversity indicators in Europe in the light of COP 2020 targets and illustrates how the indicators could be used for climate change discussions.

Keywords: sustainable forest management, biodiversity, ecosystem services, forest indicators, COP 2020 targets

1. Introduction

Predicted climate change will force us to re-evaluate present forest management practices and the use of wood. In the face of the potential negative effects of climate change on forest biodiversity, forest managers should develop adaptation measures for increasing the resilience of forests and to prevent forest damage

caused by extreme phenomena such as storms, drought, forest fires, or damage by insects. On the other hand forests play a significant role in controlling climate change by mitigation actions. Wood is a nearly the only low-energy, renewable, and carbon-neutral building material throughout its entire life-cycle. In energy production, wood biomass reduces reliance on fossil fuels, and therefore diminishes greenhouse gas emissions.

In these changing conditions, the demand for new forest products and services by various sectors of society increases continuously. Especially in the energy sector with increasing use of wood-based bioenergy, traffic sector using liquid biofuels and the construction sector using wood as environmental sound material require new products, services and up-to-date forest information. This increasing use of wood-based products should be organized by maintaining forest sustainability and forest biodiversity, as agreed to by the European Union in the climate and energy package decision in 2008. In this decision, it was agreed to develop practical applications on the sustainable use of forests for controlling the sustainable production of biomass for energy applications and for Green Public Procurement Policies of wooden products (The role of forests... CESE 626/ 2009).

There is also an increasing need to serve societies with services and forest information in cross-sectoral thematic areas such as climate change, biodiversity conservation, water, various environmental ecosystem services, manufacturing chain of wooden products, and relationships between forests and human health. Modern life styles and urbanization in the developed world, especially in countries with dense populations, have led to increased physical and mental stress of people. Also in developing countries the human health linkages to forests, needs for fibre, pharmaceutical products and clean water, or with adverse effects, such as infectious diseases (e.g. malaria) requires increased attention, especially in connection with deforestation and land-use change (Parviainen 2009, Parviainen et al. 2010).

The concept of Sustainable Forest Management (SFM) has received worldwide recognition (FAO/ITTO 2004, Forest Criteria and Indicators 2009). The criteria and indicators of SFM deliver information for decision-making, and serve as guidance for forest

¹ Ministerial Conference on the Protection of Forest in Europe, new name FOREST EUROPE

management according to societies needs. The present SFM criteria and indicator sets were compiled mainly for the purposes of the forest sector, but could provide, with some modifications, the required new information for various other sectors.

This article provides a proposal on how the existing MCPFE Criteria and Indicator set on SFM could be adjusted to serve better society's needs by combining information of the forest and other sectors and thematic areas. MCPFE forest indicators have been used to date to evaluate the status of Europe's forests. The proposal for a wide use of the MCPFE Criteria and Indicators covers following items: updating quantitative and qualitative indicators in accordance to new demands, review of overall policies and special policy areas, setting of threshold values and verification issues.

2. Criteria and Indicators for Sustainable Forest Management (SFM)

The modern concept of SFM will soon be 20 years old, but the term 'sustainable' was first mentioned and used related to sustainable yield of forest resources by the German forester von Carlowitz in 1713. Based on the UNCED Rio Declaration (1992), European countries agreed in the Ministerial Conference for the Protection of Forests in Europe (MCPFE), in Helsinki in 1993, the principles of SFM with simultaneous respect to ecological, economic, and social aspects.

SFM was defined as follows (Resolution H1, item D, MCPFE 1993):

Agreeing that, for the purposes of this resolution, 'sustainable management' means the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfil, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems.

In the follow-up work by MCPFE in 1993–2003, 35 quantitative indicators, 12 qualitative indicators, and five overall policy instruments were developed for monitoring SFM. At present the MCPFE tools include five elements for guiding and monitoring the implementation of the SFM (see Table 1 and 2). The status of Europe's forests has been evaluated and summarised according to the MCPFE tools by the State of Europe's Forests reports in 1998, 2003 and 2007 (Köhl & Rametsteiner 2007).

To date, nine intergovernmental regional sets of criteria and indicators for various forest zones and 185 countries have subscribed to the concept of SFM. These countries include MCPFE countries, Montreal Process countries of temperate and boreal forests outside Europe, ITTO (International Tropical Timber Organization) and African Timber Organization countries, Amazon region, Central American, Dry Zone Africa, Near East and Dry Forest Asia countries. This coverage provides a basis to extend the use of

Table 1 Elements included in the MCPFE tools. Source: MCPFE Internet pages at <http://www.foresteurope.org/>

1) Definitions and General Guidelines for Sustainable Management of Forests (SFM) in Europe (Resolution H1, MCPFE Helsinki 1993) with the General Guidelines for the Conservation of the Biodiversity of European Forests (Resolution H2, MCPFE Helsinki 1993)
2) Six pan-European Criteria for Sustainable Forest Management and the Improved Pan-European Indicators for Sustainable Forest Management (MCPFE Lisbon 1998, MCPFE Vienna 2003)
3) Pan-European Operational Level Guidelines (PEOLG, adopted in MCPFE Lisbon 1998)
4) Pan-European Guidelines for Afforestation and Reforestation with a special focus on the provisions of the UNFCCC (MCPFE 2008)
5) MCPFE approach to National Forest Programmes in Europe (MCPFE Vienna 2002)

SFM criteria to all continents and forest situations (FAO/ITTO 2004). For the international use of indicators, the heterogeneity and diversity of countries should be taken into account.

The SFM Indicators are an important tool in providing a balanced compendium of information on forests (ecological, economic, social and cultural) for three aspects: 1) showing long-term trends and changes in the forests, 2) integrating the forest policy goals and decisions with the measurable indicators and 3) providing a continuous basis for international comparability. Focusing on the main messages and increasing the clarity of the assessment should lead to concentration on the most important, common, easily measurable and cost-effective indicators.

While demand for various uses and reporting on the forests is increasing, the aim should be that the forest indicators information need to be reported and verified only once, at one particular monitoring cycle, and then used for many different purposes on similar definitions and concepts. The use of forest indicators in other sector's reports is important for synergies and awareness of forest issues, and also to guarantee that relevant and generally accepted terms and harmonized definitions are used. In the other sector's reports often only a few forest indicators and combined indicators are used. (Parviainen & Lier 2006).

Within the Convention on Biological Diversity (CBD) the **ecosystem approach** and **sustainable use** have been defined. There exists a tight connection between the SFM and CBD ecosystem approach definitions. It has been clarified by European Ministries of Forestry (MCPFE) and European Ministries of Environment in Europe (PEBLDS) in 2004 that the concepts of the Ecosystem Approach (EA) and Sustainable Forest Management (SFM) have the same goal to promote the conservation and management practices in forests, which are environmentally, socially, and economically sustainable. Sustainable forest management can be considered as a means of applying the ecosystem approach to forests in the pan-European region.

Table 2 Pan-European (MCPFE) Quantitative (1.1–6.11) and **Qualitative (B.1–B.12)** Criteria and Indicators for Sustainable Forest Management. Source: MCPFE 2003

Criterion: A general goal on which a judgment or decision can be based.

Indicator: A quantitative or qualitative parameter which can be assessed in relation to a criterion. It describes objectively and ambiguously the content of the criterion and provides an indication of the condition or direction over time.

The **quantitative indicators** show changes over time for each of the six criteria

The **qualitative indicators** give an overview of the policies, institutions and instruments for sustainable forest management.

Criterion 1: Forest Resources	Criterion 4: Biological Diversity
B.1 Land use and forest area and other wooded land 1.1 Forest area 1.2 Growing stock 1.3 Age structure and/or diameter distribution B.2 Carbon balance 1.4 Carbon stock	B.6 Biodiversity 4.1 Tree species composition 4.2 Regeneration 4.3 Naturalness 4.4 Introduced tree species 4.5 Deadwood 4.6 Genetic resources 4.7 Landscape pattern 4.8 Threatened forest species 4.9 Protected Forests
Criterion 2: Health and Vitality	Criterion 5: Protective Forests
B.3 Health and vitality 2.1 Deposition of air pollutants 2.2 Soil condition 2.3 Defoliation 2.4 Forest damage	B.7 Protective forests and other wooded land 5.1 Protective forests– soil, water and other ecosystem functions 5.2 Protective forests – infrastructure and managed natural resources
Criterion 3: Productive Functions	Criterion 6: Socio-Economic Functions
B.4 Production and use of wood 3.1 Increment and fellings 3.2 Roundwood 3.5 Forests under management plans B.5 Production and use of non-wood goods and services, provision of especially recreation 3.3 Non-wood goods 3.4 Services	B.8 Economic viability 6.1 Forest holdings 6.2 Contribution of forest sector to GDP 6.3 Net revenue 6.4 Expenditures for services 6.7 Wood consumption 6.8 Trade in wood 6.9 Energy from wood resources 6.10 Accessibility for recreation B.9 Employment (incl. safety and health) 6.5 Forest sector workforce 6.6 Occupational safety and health B.10 Public awareness and participation B.11 Research, training and education B.12 Cultural and spiritual values 6.11 Cultural and spiritual values

3. Interlinkages between ecological, economic and social indicators

Developing indicators of sustainable forest management requires the understanding of the broad and long-term interactions between humans and forests, including biodiversity. In our modern industrialized societies, valuation of ecosystem services needs a proper weighting of various benefits of these services. In order to make balanced decisions on forest resources and their use it necessary to know and understand these interlinkages between the ecological, economic and social dimensions of sustainable forest management. Also there is need to link the forest indicators with the indicators of other sector in the use on natural resources.

Within the European Union, a set of biodiversity indicators for agriculture, forestry, fisheries and wetlands has been developed according to the CBD focal areas during 2004–2009. An activity launched in 2005 on Streamlining European 2010 Biodiversity Indicators (SEBI 2010) aims to produce and develop consistency across global, regional, EU, and national indicators (EEA 2007, EEA 2008).

The interlinkages between the indicators can be illustrated with the development work on the European biodiversity indicators – SEBI 2010. Eight key indicators have been developed to give a balanced view on forest biodiversity and related factors in Europe. For the first set of indicators under the focal area “sustainable use”, however, only two forest specific indicators were proposed, namely 1) growing stock, increment and fellings, and 2) dead wood among the 26 indicators in Total for agriculture, forestry, fisheries and wetlands.

Background for selection of key forest indicators in European context:

For the worldwide discussion it is necessary to characterize the European forest situation, while it differs in several aspects from the situation in other continents. Despite considerable variation between European countries in relation to vegetation zones, growing conditions, tree species, organization of forestry, goals of forestry, population density, and uses of forest products, there are also considerable similarities. Common factors among European countries with respect to forestry include the ownership structure based mainly on private, family ownership, long-term human impact on forests, small area of original forest cover, and an increasing risk of forest instability due to the anthropogenic influence, air pollution, and forest stand density.

Forest area: Baseline indicator related to all other indicators developed on the basis of land cover information.

Naturalness: The degree of naturalness of forests shows the intensity and history of human interventions. Different levels of utilisation intensity are characterised not only by the

remaining forest area in the country but also by changing structures and different species communities. Composition and structure determine the functional diversity and these factors constitute aspects of biological diversity. Information on naturalness, is provided by undisturbed forests and can be used when setting up management priorities and plans and models for silvicultural planning.

Deadwood: Decaying wood as standing and fallen tree habitats are important components of biodiversity for many forest-species and recognised as potential indicators for assessing and monitoring biodiversity as well as sustainable forest management. However the amount of dead wood in protected forest areas and in multifunctional forests varies considerably.

Protected forests: The area of protected forests provides an estimate of forests protected for biodiversity in Europe. The classification system also provides quality information on protected forest areas, while classifying the areas according to the level of management.

Growing stock, increment and fellings: Balance between net annual increment and annual fellings (utilization rate) of wood highlights the sustainability of timber production over time, and the current availability and potential for future availability of timber

Gross value added: From the national perspective, the contribution of forestry, its services and products including the manufacture of wood and paper to gross domestic products indicates its macro-economic importance. This indicator can also be used for assessment of how forest management contributes to overall sustainable development, for instance to rural development and whether or not this contribution is sustainable. The number of **forest holdings in Europe** is an important social indicator, especially for sustainable development in rural areas, due to significant changes during the last decades. Private, non-industrial forest ownership is the dominant form of ownership in Western Europe. Experience shows that sustainable forest management and respect of biodiversity has been most successful when forestry is profitable. Therefore also, by promoting more nature-oriented silviculture, any expected outputs should always be economically evaluated.

Employment in the forest sector: Employment provided by forestry is an important indicator for social benefits, especially for sustainable rural development. In this connection, an adequate workforce in terms of numbers and qualifications is a critical input to SFM.

Some elements of these eight forest specific indicators are been included into the general biodiversity indicators of SEBI 2010 such as abundance and distribution of selected species, red list index for European species, species of European

interest, ecosystem coverage, habitats of European interest, nationally designated protected areas, sites designated under the EU habitats and birds Directives (Natura 2000 network), invasive alien species in Europe and fragmentation of natural and semi-natural areas. This example shows that there exist a clear difference in the indicator sets between the biodiversity approach and the sustainable forest management concept.

4. Proposal for updating and verification of the SFM indicators – as example the MCPFE tools

The set of indicators for SFM, both quantitative and qualitative could be revised with their relevance and orientation from the point of view of new requirements in better serving society needs, especially for climate change reporting, research and discussion, use and production of wooden bioenergy, forests and human health aspects, water issues, and the development of research in the immaterial, non-wood forest ecosystem services. On the other hand there is a need to show that the sustainability principles and indicators are fulfilled in the practical forest management and in the implementation of the policies. The verification needs a technical procedure and threshold values set for indicators.

4.1. Updating the indicators

There is already wide flexibility in reporting of qualitative indicators according to new policy orientations and requirements, but some new elements could be added according and parallel to the amendments to the **quantitative indicators**. Especially the orientation of policy areas needs to be revised to take into consideration for instance climate change and bioenergy issues related to other sectors, and human health and well-being issues, within the socio-economic policy area. In this connection, the interlinkages between various policy areas could be evaluated. For example, for climate change adaptation measures, a new qualitative indicator for “safeguarding forest services and wood production”, that is **forest management contingency plans** could be required.

Examples of **new quantitative indicators** that could be developed and added into the present MCPFE set:

- Green House Gas (GHG) savings in heat, electricity and biofuel/bioliquid production from wooden biomass in comparison to the fossil fuels
- Carbon storage in harvested wood products
- Nutrient fluxes and balance in areas of extraction of biomass
- Quality of groundwater as results of forest operations
- Contribution of forests to human health (mental and physical)

42. Verification

Threshold values set for indicators help to monitor the implementation of agreed measures and policies in practice. The threshold values are either scientific-based or politically based, and sometimes are mixtures of both. Targets are the results of weighing interests. At the country level, thresholds can only be defined on a general level, and they are mainly political targets.

While the conditions for forest growth and forestry vary considerably across Europe, reflecting the diversity of vegetation zones and climates, naturalness of the forests, traditional forest use, fragmentation, forest ownership structure, and stakeholder views, in practice critical quantitative levels as thresholds can only be set for a few indicators. Therefore general threshold values at the MCPFE level must be developed and modified for a practical implementation at the country level or at management unit level. These threshold levels should not be confused with political targets, which may not necessarily be the same.

The variation of recommended dead wood component in managed forests among European countries illustrates the difficulty to create a common threshold for biodiversity (see Parviainen et al 2007). The range of recommended minimum deadwood varies from 5 trees per hectare to 5 m³ per hectare. In some cases 5 % of the standing tree volume is required. There is not yet any clear scientific evidence which kind of amount should be enough for maintaining the endangered species living in deadwood habitats, especially because every single species might have various requirements. Conclusion is that threshold values are often compromises and subject for further discussions and possible conflicts.

Examples of threshold values at operational level for quantitative indicators for wood harvesting are:

- *The annual cuttings cannot exceed the annual growth on long term (qualitative and quantitative threshold)*
- *For safeguarding the biodiversity, small rare habitats must be left untouched within the managed forest area (qualitative: habitats defined and described) and a minimum of dead wood (quantitative: XX volume of trees) should be left on site*

In the European situation, relevant threshold values for indicators can be defined and set with the revised MCPFE Pan-European Operational Level Guidelines (PEOLG) for SFM (see Table 1). The guidelines could be updated also in relation to the new requirement, and possible new indicators added.

Verification means the procedure by which it can be shown that the sustainability principles are fulfilled. Verification procedures are typically used in forest certification at forest management unit level (PEFC 2010, FSC 2010). If the SFM criteria and indicators are made verifiable at the country level, then these sets can be used for third party assessment and

monitoring in a similar manner to forest certification. A verification process for SFM indicator sets is needed for EU FLEGT (Forest Law Enforcement, Governance and Trade) licensing scheme for imports of timber into the European Union, Public Wood Procurement Policy and of REDD (Reduction Emissions from Deforestation and Forest Degradation in Developing Countries) instrument on the country level.

The MCPFE tools (see Table 1) include, in principle, the verification requirements, but the tools are not formulated technically verifiable. A verification process applies the verification of both qualitative and quantitative indicators. However, the MCPFE tools can provide, with some modifications, necessary elements for verification. For instance the MCPFE approach to National Forest Programmes deals with procedural elements of policy, and these elements can be integrated closely with quantitative indicators. The National Forest Programmes in Europe, and the qualitative indicators (overall policies, and relevant indicators) could be studied and modified for verification purposes.

The verification of quantitative indicators can be undertaken on the basis of modified MCPFE PEOLG guidelines with relevant threshold values.

The qualitative procedural elements of the MCPFE tools are broad and general in their coverage of social dimensions. In the assessment of forest certification systems, significant emphasis has been put on decision making bodies and appeal procedures, which implies the need to strengthen these social dimensions. The dimensions related to participation and good governance have not been developed to the indicator level, although for instance the National Forest Programmes have been normally created with a wide and representative stakeholder panel.

5. Forest indicators serving other sectors and thematic areas – for example, climate change discussions

The SFM indicators create a reservoir, from which the most relevant indicators can be selected for various purposes on the basis of a balanced compendium of information on SFM. The SEBI 2010 work in Europe (see pages 3 and 4) is one example of how these indicators can be used for the evaluation of biodiversity.

On a similar manner, the SFM indicator reservoir could provide information for climate change discussions related to forests. The following key quantitative indicators could be selected for climate change:

- *changes in forest area and consequently changes in carbon sequestration,*
- *proportion of strictly protected area/ forest area in wood production or multifunctional purposes,*
- *the ratio between annual harvest and growth of forest (data from which carbon sequestration*

capacity and changes can be derived),

- carbon sequestration in harvested wood products,
- substitution aspects as GHG saving by the production of liquid biofuels, heat and electricity,
- minimizing the harmful environmental effects of biomass extraction, and
- information on which kinds of measures have been developed in the countries for forest adaptation: in addition to management such as contingency plans with mapping of risk areas in the case of extreme weather phenomena.

To make this forest information and reporting attractive for public audiences and decision-makers it is necessary to reduce description of indicators to a minimum and add some illustrative aspects, such as maps, photos, simple figures and graphs, and concentrate on a few key indicators (Parviainen & Lier 2006). A brochure with a reduced selected set of indicators for special thematic areas, showing graphically the main characteristics on the status of forests, has often been positively received by high level policy makers. Internet based communications is a practical tool for the dissemination of information regarding forest sustainability.

6. Key forest biodiversity indicators in Europe in the light of COP 2020 targets

6.1. Status of forest biodiversity in Europe

The MCPFE forest indicators illustrate forest biodiversity in Europe as follows (Köhl & Rametsteiner 2007):

European forests have been functioning for several decades now as carbon sinks because their annual growth has exceeded fellings, thus helping to slow the build-up of carbon dioxide in the atmosphere. This means that the forest utilisation rate, or ratio of felling to growth, was over the last 40 years less than 60% on average within the European forest area (Figure 1 and Figure 2). It is estimated that the annual growth of European forests sequester approx. 10% of Europe's annual carbon dioxide emissions (Nabuurs et al. 2003).

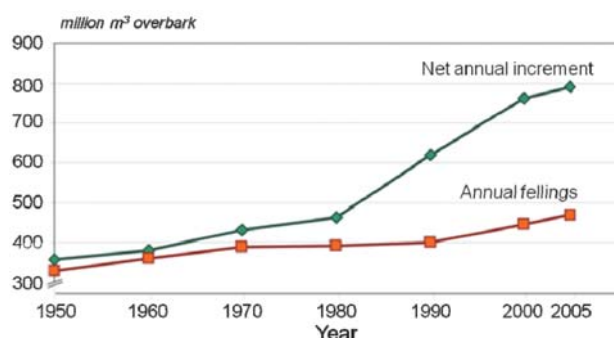


Fig. 1 Forest utilization rate on forest available for wood supply expressed by the relation between annual growth and cuttings in the European countries in 1950–2005. Sources: Kuusela 1994 & MCPFE State of Europe's Forests 2007

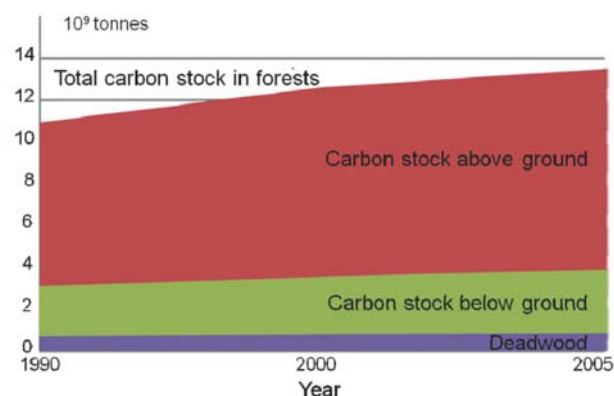


Fig. 2 Trends in carbon stocks in forest biomass and deadwood in the European countries in 1990–2005 excluding Russia. Source: MCPFE State of Europe's Forests 2007

Forests cover 31% of Europe's land area, but long-term human intervention has change the structure of European forests. Most European forests (87%) are classified as semi-natural forests (Fig. 3). The forest area dominated by introduced tree species is around 4%, and the total area is not increasing (Parviainen et al. 2007).

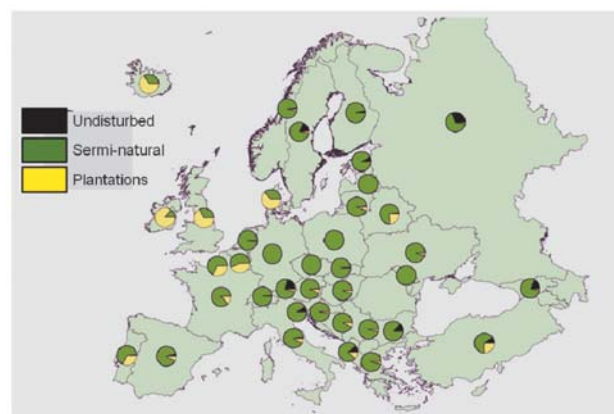


Fig. 3. Share of the classes of naturalness (undisturbed by man, semi-natural forests and plantations) as % of the forest area in 2005 in the European countries. Source: MCPFE State of Europe's Forests 2007.

About 8% of Europe's forests are protected with the main objective of conservation of biodiversity and another 10% with the main objective of conserving landscapes and specific natural elements (Fig. 4). The area of protected areas has increased considerably over the last 20 years. Forest management practices have changed in ways that promote the conservation and enhancement of biological diversity, notably through an increased use of natural regeneration and more mixed species stands.

About 70% of Europe's forests are dominated by mixed forests consisting of two or several species. Nearly 50% are regenerated by natural means, and that amount is increasing. Measures are also being taken to encourage deadwood accumulation. The average amount of deadwood is about 10 m³/ha, but

this varies depending on the growing stock volume by forest types and vegetation zones.

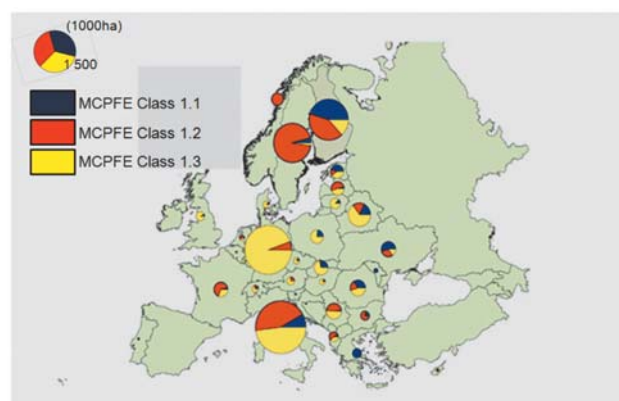


Fig. 4 Protected forest areas (as % of the forest area) for biodiversity in the European countries in 2005 according to the MCPFE classes: 1.1–1.3 (1.1. no interventions, 1.2 minimal interventions, 1.3 active management for biodiversity). Source: MCPFE State of Europe's Forests 2007.

62. COP 2020 target on indicator forest protection

The indicator on the area of protected forest areas is a practical tool with which to monitor the change.

European forest biodiversity has become a very complex and varied issue because of the intensive historical use of forests, the specific small scale of the private ownership structure, and the fragmentation of forests within the landscape caused by other land uses. Therefore protected forests areas are often small, with most located in fragmented landscapes and protected with various management options and regimes. Based on these facts, the MCPFE Assessment Guidelines for Protected and Protective Forest and Other Wooded Land (MCPFE classes) were created in 2001–2003 especially for European conditions.

The results of this UNECE/FAO data collection process, according to the MCPFE PFA guidelines in 2003 and 2007, show that this classification is workable and provides a comprehensive and versatile overview of the European situation. Thus national networks of protected forest areas should not be seen in isolation but as a part of an overall forest management and biodiversity strategy.

In Europe, the main emphasis in protection for biodiversity is on active management. The share of protected forests for biodiversity with no active intervention (strict protection) is small, 0.9%. This is logical because nearly all the rare and vulnerable forests in Europe are already protected. The ideal non-intervention, strict protection concept, is not a realistic scenario in Europe. Natura-2000 conservation network is a special tool within the European Union. Natura-2000 network aims to maintain species and habitats in addition to the protected areas also in multifunctional forests when appropriate. Harvests are allowed if the favourable protection status in habitats designated in EU Directives is maintained. The CBD

SBSTTA targets 2020 should be set by taking into account the local forest conditions and all legal instruments available (legal protection areas, voluntary legal contracts and tenders, Natura-2000 areas and close to nature management) **as integrated protection approach.**

63. COP 2020 target on resilience under climate change issues and biodiversity

According to the European Economic and Social Committee (The role of forests... CESE 626/ 2009), in the face of the potential negative effects of climate change, EU Member States should develop forest management contingency plans to prevent forest damage caused by extreme phenomena (storms, drought, forest fires, damage by insects) and for remedying the effects of such damage, in addition to increasing information about the importance of continuous good forest management.

The importance of natural forests as carbon stores and as preserves of biodiversity must be ensured. However there is an important difference between commercial forests and natural forests in terms of carbon sequestration. From the perspective of climate mitigation, natural forests in their "end state" are carbon stores, in which carbon sequestration through the growth of biomass and carbon release through the decay of biomass are in equilibrium. Whereas commercial forests act as a carbon pump as they are constantly developing new and additional carbon sequestration capacity due to the harvesting of timber and timber use for wooden products and for substitution of fossil fuels by bioenergy.

Based on continuous forest monitoring in Europe, wood resources allow a considerable expansion in the use of wood for construction and also for forest bioenergy purposes, provided that close to nature forest management is used and the harmful environmental effects are minimized.

7. Conclusions

The information gathered by indicators regarding the forest sustainability is important and unique providing a balanced compendium of information. The main users of indicators and national reports based on these are governmental officials and scientists who need the data for international sustainability reports, tools for forest policy and strategies, public information on forests, impact on forest research and other research initiatives. The examples of two selected indicators presented in the article, namely protected forests and carbon sequestration illustrate the possibilities to fulfil the political requirements of COP 2020 targets.

While demand for forest information and various reporting is increasing, the use of forest indicators in other sector's reports is very important for synergies and awareness of forest issues. Currently, the forest sustainability indicator sets are not flexible enough to provide the required data for example, climate change

impact, adaptation or mitigation such as wood based bioenergy. These requirements call for customer oriented forest data collection and reporting, where the information should be however proportional to the other forest indicators. There is a need to communicate with other sectors which forest indicators could be selected in order to give a balanced and focused view on the forests and its uses.

The analysis and examples provided in this article shows how the present MCPFE criteria and indicator set and MCPFE tools could be adjusted to serve better society's needs by combining information of the forest and other sectors and related thematic areas. The proposal for a wider use of the MCPFE Criteria and Indicators covers the items: updating quantitative and qualitative indicators in accordance to new demands, review of overall policies and special policy areas, setting of threshold values, and creating a verification procedure. The review of MCPFE tools and implementation work needs the political decision in the Ministerial Conference.

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How scientists can contribute to the CBD and the post-2010 targets: Challenges in raising public awareness and lessons learned from NGO campaigns

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Abstract

The Convention on Biological Biodiversity is based on science and as such the scientific community has played a critical role in shaping the Convention. They have monitored the status and trends of biodiversity through the development of indicators and have identified new and emerging issues for the parties to the Convention to address. However more recently, with the expiration of the 2010 Biodiversity Target, the scientific community has broadened its focus and began to consider ways and means to monitor not only biodiversity but also how countries are implementing the Convention. A similar role has also been played by non-governmental organizations (NGOs), which have raised awareness of many biodiversity issues through their campaigns and ground level actions. In this article, case studies of Japanese scientists in developing the science-policy interface are examined and reviewed, with several suggestions on how this relationship could be improved. Lessons learned from studies of the NGOs are highlighted. In particular, issues related to urban forestry are examined.

Key Words: Public Awareness, Science-Policy Interface

1. Introduction

Scientists and non-governmental organizations (NGOs) have played an important role in bringing biodiversity issues to the attention of the general public. For example, both scientists and NGOs helped to move the destruction of tropical rainforests onto the international political agenda during the 1980s. NGOs in particular, have often been successful in capturing the attention of the media and politicians through their “radical” forms of communication such as mass demonstrations, in addition to conventional information distribution. Further scientists and NGOs played an important role in providing the foundation for the creation of an international biodiversity convention. Scientific community identified the problem and then NGOs helped to place it on the international agenda. The role of scientists and NGOs differ; scientists analyze the relationships between the elements of the biodiversity and the casual

relationships of their changes, or the driving forces behind the changes. NGOs are more active in agenda setting and mobilize the scientific findings but their framework or foundations are not necessarily on a rigorous scientific basis.

While the input of the scientific community was crucial in creating the Convention, more recently some scientists argue that there is room for improvements, especially in the dialogue between science and policy, which has not been necessarily effective in helping to implement the Convention (cf. Siebenhüner, 2007; Koetz et al., 2008). Therefore, an increasingly urgent task for these two communities is to further develop the link between science and policy, the so-called “science-policy” interface (SPI) for biodiversity related issues.

There are existing studies on how various NGOs have different strategies. For example, Wapner (1994) pointed out that Greenpeace acted relatively independently while Friend of the Earth tried to accommodate each social contexts with what he called “political internationalism.” In other words, Friends of the Earth oriented their tones and actions towards local contexts, while Greenpeace acted rather uniformly with common causes. Similar discussions are found for the whaling dispute in Japan between the government and WWF-Japan, which expressed different messages as compared to the tones of the international community. There is also a detailed case study on specific communications strategies of NGOs by analysing how they construct their story for the general public through media (DeLuca, 1999). A comparative analysis of NGO’s role in Japanese and German examples for the policy formation of the Chlorofluorocarbons (CFCs) is analysed in Matsumoto (2007) (Matsumoto, 2007).

So far, discussions on SPI has largely focused on formal organizational aspects (cf. Kohsaka and Minohara, 2009 for examples in CBD and CITES). In this paper, the SPI and its implications for raising public awareness are discussed; the local politics of urban forestry in Japan are analyzed and the role of scientists and NGOs in resource management and raising public awareness are critically reviewed in a general sense.

2. Biodiversity awareness in Japan

2.1 General public

Biodiversity awareness raising has been a challenging task in Japan. According to the third National Biodiversity Strategies and Action Plan (NBSAP) published by the Ministry of Environment, only 30.2% of the public has heard the term “biodiversity” before, while only 6.5% of the population knew the meaning of the term (Ministry of Environment, 2008; 216). A more recent study undertaken by the Cabinet Office of the Government of Japan in 2009, indicated that the 12.8% knew the meaning of biodiversity, 23.6% reported having heard the term but did not know the meaning,” and 61.5% answered that they had never heard the term. The Conference of the Parties (COP) to the CBD was less known and a marginal 3.8% knew the term and 9.3% had heard the term before. These results indicate that the level of awareness surrounding biodiversity is relatively low. For this reason, in its third NBSAP, the government of Japan set the goal of raising awareness from 30.2% to 50%, and from 6.5% to 15% respectively by March 2011. In order to accomplish this, there will be a need for further and more effective communications between the public and policy makers, scientists and NGOs.

Similar conclusions are made in case of European citizens that 35% of the European citizens knew what the term “biodiversity” meant and another 30% had

heard the term before (European Commission, 2007). The ratio that knew the meaning of term is higher than the public in Japan, although the ratio that “had heard the term” is close to 30%. It was also pointed out that the general public understood biodiversity loss mostly as a species-focused concept or as a concept related to changes in natural habitats.

2.2 Business sector

A survey conducted in 2006 (Ministry of Environment, 2006) found that 73% of 2774 the listed and non-listed companies answered that “biodiversity conservation is important but less relevant to the operation of the company.” The ratio of the companies that responded that biodiversity conservation was “relevant and a priority” were in the minority with 11.6%. The proportion of companies indicating that biodiversity was both important and a priority was highest amongst companies in the utility sector (electronic power and gas). Companies related to land use, such as construction and real estate sector, also had a higher proportion of recognizing the importance of biodiversity and that it was a priority, as compared to the service sector—such as banks and securities, wholesale, and retail—which had the lowest recognition.

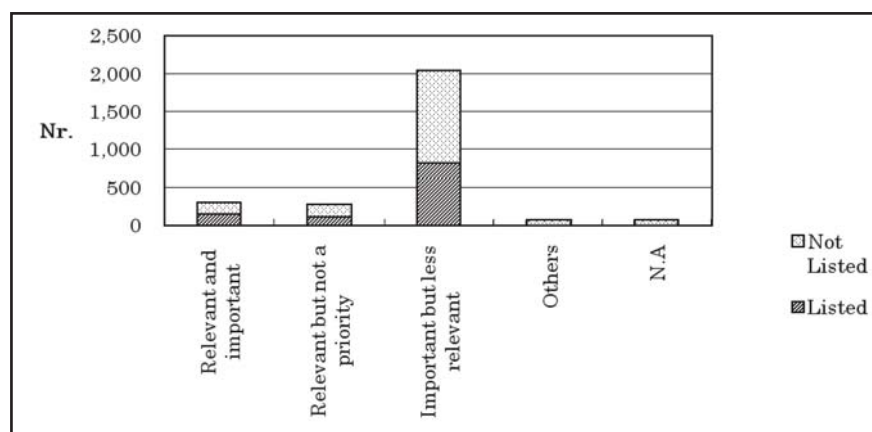


Fig. 1 Biodiversity Conservation and Corporate Activities in 2006

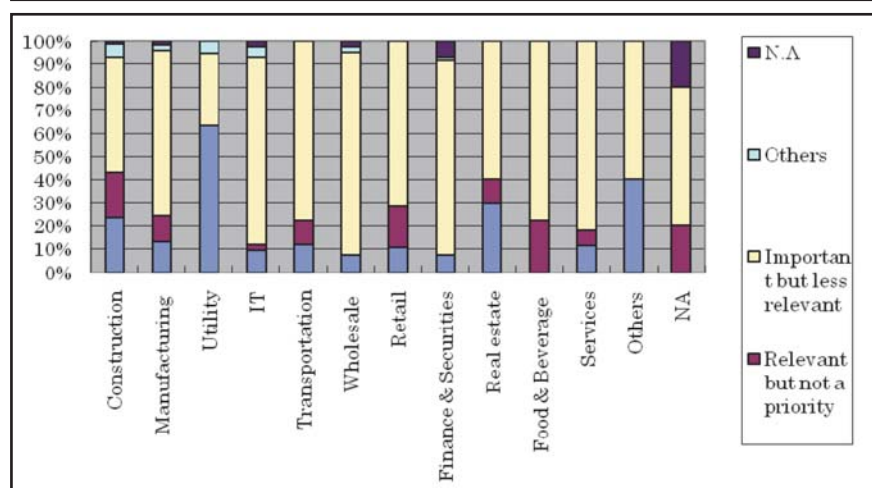


Fig. 2 Biodiversity Conservation and Corporate Activities by Sectors in 2006

The Nippon Keidanren business association, one of the largest economic organizations in Japan, conducted an additional study on the business sector's awareness of biodiversity issues in 2009. The survey was conducted as part of the preparation of the associations Declaration on Biodiversity. A questionnaire was distributed to major member companies of the Nature Conservation Committee in Keidanren. 145 companies responded and the results were analyzed to review what the companies regarded as future tasks and risks related to their operations and biodiversity (Kohsaka and Tokuyama, 2009).

In the results relating to the companies perception of future tasks for the conservation of biodiversity, the highest ranked activity was associated with securing resource savings, while the second was ensuring safe procurement for supply chains. The third highest ranked activity was the development and innovation for environmental technology. The majority of the responding companies were manufactures, thus the activities regarded as future tasks were more related to resource saving and technological innovation. Issues pertaining to the CBD, such as activities related to the fair and equitable sharing of genetic resources, biosafety and the exclusion of invasive species, were of relatively low interest to the respondents (with the exception of a few retail, trading, cosmetics or food & beverage companies that responded that these issues were a priority).

In terms of risk perception, across the sectors companies perceived that the greatest threat to their operations was the risk of losing their reputations. Nearly one third of the companies responded that their reputations were connected to CSR (Corporate Social Responsibility) and the greatest threat to it was if biodiversity-related activities were neglected in their operations. This data indicates that companies' operational risk—such as disturbance of supply chain or lack of water supplies—are not perceived as being related to biodiversity issues, but rather remain conceptualized as an open system based on infinite resources. Thus, ecosystem services (that are an intricate aspect of biodiversity) and their linkage to sustainable management need to be highlighted and awareness of its importance needs to be raised.

2.3 Scientists

As mentioned earlier, scientists played an important role in raising awareness about biodiversity issues and the need to take action in the international political agenda in the 1970s and the 1980s. Currently there are a diversity and broad range of projects undertaken by the scientific community in the field of biodiversity. The examples focused upon in this paper are the Millennium Ecosystem Assessment (MEA), which involved some 1500 scientists and the Global Biodiversity Outlook (GBO), which assess and reports global trends on biodiversity and is largely based on data and resources from scientists, including the MEA,

as well as from NGOs. In an on-going project, economists and non-governmental organization, such as IUCN, are contributing to the Economics of Ecosystems and Biodiversity (TEEB) that is being compiled by UNEP and the European Commission, in addition to other governmental bodies.

There is less data relating to scientists' awareness of biodiversity. In Japan, certain aspects of awareness can be observed based on the questionnaire Comprehensive Analysis of Science and Technology Benchmarking and Foresight (CASTBF) conducted by the National Institute of Science and Technology Policy (NISTEP) that is affiliated with the MEXT (Ministry of Education, Culture, Sports, Science and Technology). CASTBF has been conducted by MEXT since the early 1970s with benchmarking taking place every 5 years as reports are published (NISTEP, 2005). Currently, the 9th Survey is being conducted for publication in the year 2010. From the tentative results of the 8th Survey Group No.8 (as of January 2010), which focused on environmental science and technologies, awareness of Japanese scientists can be characterized; the most frequently chosen options, "risk assessment/management/communication," were marked as important for both the world and Japan. As world-wide priorities, "pollution prevention in air, water, soil" and "circularize use of water resource" were highlighted. Japanese priorities included "urban and rural environment (regional environment)".

Technical challenges for biodiversity and ecosystems can be characterized by comparing them to climate change related ones. Biodiversity tasks were ranked as relatively high priorities for Japanese scientists. In other words, technical challenges for Japanese scientists are perceived as both a priority and as a domestic and international commitment, with the domestic portion being larger. Most of the scientists regarded technical feasibility of environmental technology (including biodiversity related topics) as a major challenge for the near future or between 2015 and 2020.

2.4 NGOs

To date, there is no comprehensive data or statistics relating to awareness raising activities of NGOs in Japan. The data is rather limited to the types of activities and number of NGOs in the field of environment. For example, a large scale list of the Japanese NGOs is compiled by the Environmental Restoration and Conservation Agency of Japan. The organization is affiliated with the Ministry of Environment. However, there are very limited resources available for the effectiveness or type of activities by the various Japanese environmental NGOs. Staff members often express needs and demands for further tools and measures to strengthen the outreach of the conservation activities by the Japanese NGOs.

In a review of the literature review there are many

individual analyses of campaigns and media in both Europe and the United States (mostly qualitative analysis). However, many of these lack ways or means forward. For example, while Josiah (2001) outlines the challenges in outreach activities, he does not quantify them, nor does he suggest any measurable indicators, as it is difficult to quantify the effects of NGO campaigns. Taking this into consideration, there are several lessons to be learned for scientists from the past experiences of NGOs. The following statements are brought forth by the authors based on informal interviews with Japanese NGOs and stakeholders:

- Timing of press releases and campaigns are critical. In certain cases, the NGOs prepare for campaigns even before the event takes place.
- A mediator is needed to bridge the gap between the scientific information and the general public.

A successful example in the case of biodiversity was the press-release of updated trends of the Living Planet Index by the WWF (World Wide Fund for Nature) at the event of CBD COP9 in Bonn, Germany. As the data was released to the media, at the early stage of the event (during the first week of the COP-MOP), the outcome was taken up by the different media. Another case in point is the flooding in Europe: because the NGOs in Europe anticipated the event in advance they prepared materials—basically campaign banners for climate change were prepared before the flooding happened in the summer of 2007. Thus the boats were ready and the prepared banners were presented to the media when the flood took place and as TV crews broadcasting.

As for the second point of the mediator, the mind-set of scientists need to be changed. Often, the goal of scientists is to publish papers in scientific journals. However, it is not enough to publish scientific findings in order to reach policy makers and the general public (cf. Kohsaka, 2008 for cases in forestry sciences). This is also a point raised by Hannigan (1995) using the example of acid rain and a scientist in Northern Europe who propagated the phenomenon. A scientist played a key role in translating the findings of the scientific data into stories understandable for the general public.

Scientists can grow from these experiences and learn the importance of providing their findings in a timely manner to international processes without sacrificing scientific credibility (in the process, collaboration of peer reviewers are necessary for timely decision).

3. Case Study in urban forestry in Aichi, Nagoya

3.1 Backgrounds

In this section, as an illustrative example of the local level, two disputes in Nagoya are described to highlight the role of scientists in the decision making

process for natural resources and environmental management in urban forestry. In Nagoya, the debate on urban forests began in the late 1980s when the City was running a bid for the Olympic Games (which in the end was to be held in Seoul, Korea). The turning point in the administrative process involved public participation with scientists playing a key role. Nagoya City went through intensive discussions on balancing development and the environment. During this period, the City nominated a number of areas, including the remaining urban forests, to be developed as stadiums and sports facilities. Certain local groups initiated protests against such development heating the debate. But in the end, the City of Seoul won the nomination and some of the areas that were planned to be developed now remain as green areas or forests. Amongst others, Higashiyama, the largest urban forest area in the City, remained mostly unchanged.

3.2 Two Projects of Urban Reforestation

Following these debates, a number of sites had raised discussions as to ways and means to accommodate various claims and interests. The demand for the urban forests varied from environmental education, recreation, alleviation of heat waves, to more developmental purposes. In the following section, two projects of urban forests are reviewed with the focus on changes in the public participations and integration of various stakeholders in the region.

As examples, experiences of two different urban forest projects are highlighted. These two projects urban forests that were contrasting in their experiences of public participation and the examples are in Idaka and Aioi area.

The projects are both initiated by the city and the purpose were both reforestation with citizens participation in a similar time period. The Idaka project went through a heavy public debate while project in Aioi was less controversial.

3.2.1 Idaka Project

The first case, reforestation projects in Idaka was one of the most disputed urban forests in the region. Different stakeholders and citizen groups made different claims for the use and services at the Idaka.

Youth groups requested the grasses to be removed and asked for the streets with clearer views for educational purposes to be constructed. Other groups focusing on activities on nature observation claimed that the site to remain unchanged. Resident groups asked the grasses not to be removed because they feared that the insects will spread if changes in grasslands to take place.

The staff members from the environmental division at the time were not well prepared for such disputes and they lack human resources for procedural and technical advice. It was difficult for the local authorities to accommodate these claims as they had diverse dimensions interests. In other words, the

claims often posed trade-off for the same site.

Meetings with interested groups were organized that were frequently interrupted with different claims. Scholars with ecological backgrounds served as the mediator for this dialogue. The scientific knowledge had contributed to the awareness that anthropogenic influences are necessary in maintaining the urban forests. These efforts had changed the perception that leaving the area unchanged does not always lead to the conservation of the area.

At the later stage, the different groups agreed on having a one focal point to liaise with the local administration. Thus, the Idaka project experienced disputes but had found ways of decision making in participatory manner of hosting dialogues.

3.2.2 Aioi Project

The reforestation program at the Aioi area was less controversial. The area is largely regarded as one of the first successful reforested area with the initiative and the participation of the citizens. It started with 20 ha of the reforestation projects in 1998 and Aioi currently has 123 ha of urban forests.

The City of Nagoya funded 8 public events with 400 participants under the framework of the project. The events were run by the grassroots local organizations. At the initial stage, it was a popular sites for nature observations. It gradually opened up for other groups and has become a site for environmental education for children. The use of urban forests for multiple purposes contributed for the acceptance of the project in the area.

Based on the experiences from the Idaka, the key factors of success in the Aioi area as having a common understanding of the problems. The major ones are; (i) under-use of the urban forest areas and (ii) communication of citizen groups with the local authority.

One of the reason that the forests were not visited nor appreciated was because they were not maintained. As a result, the forests became even more difficult to approach. The vicious circle of less visitors lead to the forests to be associated with illegal dumping and other crimes. The actual flow of the volunteers and maintenance of the landscape gradually lead to the return of the visitors to the sites. The lesson learned both for the local authorities and the civil society is that the urban forests need to be maintained by human labor and visited. Otherwise, the ecosystem service will decrease and will lead to unwanted results.

Regarding the second point, the relationship between local authority (i.e. city) and the civil society shifted as well. The improved communication and relationship between the civil society and the local authorities was recognized as key area in this project from the beginning. The topic considered as one of the common challenges. Trusts between the two were gradually formed. In the process of the dialogue, the local authority came to realize the knowledge and

expertise adjusted to the individual local sites accumulated in the local civil societies. Their knowledge and expertise were developed mainly through voluntary activities in nature watching and environmental education. Such accumulated expertise is increasingly mobilized for maintaining the urban forests.

3.3 Experiences from Nagoya and the role of scientists in awareness raising

From the experience of the Nagoya, the public participations were the key issues in two cases. Scientists functioned as the mediator of the dispute in one case while collaboration of local municipality and the NGOs (or citizens group) were more central in the other case. Changes in institutional arrangements for integrating social values and public participation were the one of the key factors. The integration of ecological knowledge into the process and the improved communication was identified as key element for the so-called "adaptive co-management" (Elmqvist et al., 2008).

After the two projects, platform with 28 groups composed of civil society were formed. The organization was titled "Liaison for Nagoya Forest Creation" (Nagoya Moridukuri Partnership Renrakukai) supported by the City. Scientists are less active in these activities and most of the activities are organized by the NGOs (although some retired scientists are participating).

Facing the financial constraints, the City operated flexibility in conserving the urban forest lands by borrowing the lands rather than buying up the lands. This enabled larger lands to be avoided from the development. Following the similar vein, the City is collaborating with local banks and regional banks to provide incentives for the house owners by giving preferred rates to green their properties.

Further steps are being explored to delegate the citizens group to officially maintain the areas under urban parks. These include maintenance of lawns, bamboo trees, and riversides which were formerly limited to local authorities.

There are legal changes in the directives that underpin this delegation. Based on the overall framework of the "Nagoya Basic Plan for a Greener City of Nagoya" (Kihon-Keikaku) developed in 2000, an additional legal instrument called the Nagoya City Green Environment Promotion Ordinance (Midori-no-Machizukuri) came into force in 2004. The Ordinance had listed the (re-)creation of forests as one of the steps forward. It also outlined the necessity to collaborate with the civil society which enabled certain delegations of the maintenance activities to the local civil society.

There are two emerging moves. The networks of municipality are linking to experiences in other cities. Participatory methods are developed in observing the invasive alien species in the urban area. A concept of

networking through sharing the experiences and database are being explored, for example at the city level. The second issue is related to economic incentives, and regulation is another topic that is emerging. As the private land has most potential in enhancing the biodiversity, economic incentives of lower loans by private banks are introduced in collaboration with the City of Nagoya (cf. Kohsaka, 2009). Besides the legislative changes, the elements of the institutional change in communication and the informal networks of sharing experiences are increasingly becoming important.

In summary, the scientists played a critical role in a local dispute by serving as facilitator with objective knowledge. Currently, projects are managed and maintained mainly by the NGOs. Scientists contributed to form a dialogue interface amongst groups of citizens with different activities and also with the local municipalities of the governments. The participation of the scientists promoted dialogues based on facts, forming a sense of trusts amongst the participants.

What are the lessons to be learned from the experiences of Nagoya? Scientists, civil groups and government officials, including local ones, need to be aware of the differences and timing of the technological feasibility and social processes. As in the statistical data by the scientists in section II, the scientists tend to be optimistic about the technical and scientific feasibility that most ecological challenges can be addressed from technical or technological point of views in near future. It is to be born in mind that technological solutions or just having a institution set up does not necessarily translate into the implementation. The time-lag of such technical and technological feasibility and social feasibility, when society with various stakeholders participation in the dialogues and move for implements are different, sometimes taking longer period of time for consents or agreements. In the questionnaire the period that the new technological feasibility and social one lagged only 5-7 years in the field of ecological restorations, the social dialogues may take longer period or require further commitment by the scientists.

4. Concluding Remarks

Based on the review above, there are several areas identified for further mainstreaming of the biodiversity in the society by the Japanese scientists. As clear from the data, the term “biodiversity” (and presumably “ecosystem services”) are not well understood by the general public in Japan. The political event of COP 10 is less known and these are urgent tasks for both scientists and NGOs to make the public aware of the term and political processes. These remain as the major challenge for the scientists in Japan.

Lessons from the NGO indicate that the timing of the

inputs and release of the findings are critical. Based on these experiences, it is critical that the scientists provide their inputs to the international processes and to the media at these events, such as at the SBSTTA. In addition, scientists need to mediate the findings and information in an understandable way in order to raise awareness for the CBD and post 2010 target. As in the case of the Nagoya, scientists can serve as the moderator in case of a dispute of natural resources in local contexts.

The scenario presented and endorsed by the scientists such as in the MEAs and GBO2 (and potentially GBO3 in future) had been one of the most widely quoted sources and hence contributed to the mainstreaming of the biodiversity. These processes are most likely to be useful for the post 2010 processes. The NGOs has contributed in identifying alarming trends and making the general public aware of the trends. Alternatively, most of the scientists regard the technical feasibility of the biodiversity related topics to be around 2020, according to the survey of NISTEP. The optimistic outlook of the technical matters (both domestically and globally) and the alarming trends of the biodiversity status seem to coexist in the discourse. Further researches and dialogue is needed to see how these two views will be scaled.

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Global Forest Biodiversity Targets and the Need for Scientific Monitoring

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Abstract

This paper provides an overview of the policy context for global forest biodiversity targets and discusses the need for scientific monitoring. The global community has set several policy targets at the multilateral level which are directly or indirectly related to the conservation and sustainable use of forest biodiversity. Such targets currently include the Strategic Plan and the 2010 Biodiversity Target set by the Parties to the Convention on Biological Diversity (CBD), the four Global Objectives on Forests agreed upon under the auspices of the United Nations Forum on Forests (UNFF), and the Millennium Development Goals. In addition, several new targets for the time beyond 2010 have been suggested. The monitoring of these targets requires sufficient scientific data and knowledge. Despite the improvements in data availability and criteria and indicators to measure the achievement of the targets, further research and development needs exist. In relation to forest biodiversity, these include further improving the monitoring of forest biodiversity at the national level (in particular in developing countries), using easy-to-use yet robust methods and the best scientific analysis available; refining and/or operationalizing the definitions of certain terms, such as forest degradation and the classification of forest types; analyzing the patterns of success or failure at the national and local level in reducing or halting the trend of forest biodiversity loss; and providing early indications of the feasibility of medium- or long-term political targets related to forest biodiversity, such as the potential for large-scale forest landscape restoration.

Keywords: Forest biodiversity, biodiversity targets, monitoring, CBD, global forest policy

Introduction

The global community has set several policy targets at the multilateral level which are directly or indirectly related to the conservation and sustainable use of forest biodiversity. Such targets currently include the Strategic Plan, set by the Parties to the Convention on Biological Diversity (CBD), and the 2010 Biodiversity Target, which aimed to achieve by 2010 a significant reduction of the current rate of

biodiversity loss at the global, regional and national level¹. Other current forest-related international targets include the four Global Objectives on Forests agreed upon under the auspices of the United Nations Forum on Forests (UNFF), and the Millennium Development Goals. In addition, several new targets for the time beyond 2010 have been suggested. Despite the improvements in data availability and criteria and indicators to measure the achievement of these targets, further research and development needs exist. The monitoring of these targets requires scientific input to develop methods that are robust and can be used to generate accurate data for indicators. Scientific input is also needed for the analysis and assessment of the indicator data collected. This paper will give an overview of the policy context for global forest biodiversity targets and elaborate on the need for scientific monitoring.

The CBD Draft Strategic Plan for the Post-2010 Period

In 2010, the CBD will revise and update its Strategic Plan. A draft of the plan will be discussed at the fourteenth meeting of the Convention's Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA 14) and the third meeting of its Working Group on the Review of the Implementation (WGRI 3) in Nairobi, Kenya, in May, and then considered at the tenth meeting of the Conference of the Parties (COP 10) in Nagoya, Japan, in October. The draft plan contains several targets that are directly relevant to forests. By 2020, it aims to achieve the following forest-related targets:

- "The loss and degradation of forests and other natural habitats is halved" (Draft Target 5)

This target could be achieved through improvements in production efficiency and land use planning and by recognizing the economic value of the ecosystem services provided by forests. From a CBD perspective, emphasis should be on preventing

¹ In its Global Biodiversity Outlook 3, an assessment of the current state of biodiversity, the CBD concludes that the 2010 biodiversity target has not been met. None of the sub-targets was achieved globally, although some were partially or locally met (Secretariat of the CBD 2010).

loss of primary forests and other habitats that are of high-biodiversity value. This target requires an improved operational definition and monitoring capacity for degraded forests.

- *“All areas under agriculture, aquaculture and forestry are managed sustainably” (Draft Target 7)*

Regarding forests, useful tools to achieve this target can be the criteria for the sustainable management of forests that have been adopted by the forest sector (ITTO/IUCN 2009). Moreover, important lessons can be learnt from the customary use of forest biodiversity by indigenous and local communities. Forest certification schemes can play an important role in achieving this target by providing incentives for a more sustainable use of forest resources.

- *“At least 15% of land and sea areas, including the areas of particular importance for biodiversity, have been protected through representative networks of effectively managed protected areas and other means, and integrated into the wider land- and seascape” (Draft Target 11)*

Particular emphasis in this context is needed to effectively protect tropical and boreal forests, and in particular primary forests which are rich in biodiversity and in carbon. There is also a need to enhance ecological connectivity, through effective national and regional systems of protected areas. Protected forest areas should be established and managed in close collaboration with indigenous and local communities, where appropriate.

- *“The contribution of biodiversity to ecosystem resilience and to carbon storage and sequestration are enhanced, through conservation and restoration, including restoration of at least 15% of degraded forests, thereby contributing to climate change mitigation and adaptation and combating desertification” (Draft Target 15)*

Appropriate incentive schemes (such as “Reducing Emissions from Deforestation and Forest Degradation – REDD-plus”) could enhance the conservation, restoration and sustainable management of forests and, with appropriate safeguards, could deliver substantial benefits for biodiversity and local livelihoods. Monitoring, as it is currently developed, will have to be an integral part of these incentive schemes. Moreover, recent developments, such as commitments of countries under the UN Framework Convention on Climate Change (UNFCCC) Copenhagen Accord, open new opportunities to link efforts for the conservation and sustainable use of forest biodiversity with climate change mitigation and adaptation measures. Forest landscape restoration, as promoted, among others, by the Global Partnership on Forest Landscape Restoration (www.ideastransformlandscapes.org)

offers the tools to achieve synergies between international commitments under the Rio Conventions, and the UN Forum on Forests. Key challenges for the monitoring and assessment of incentive schemes will be, among others, defining clear indicators for forest degradation and criteria for when a degraded forest has been sufficiently recovered.

Other forest-related international targets

The CBD draft Strategic Plan is only one of several targets that have been set by the global community that are directly or indirectly related to the conservation and sustainable use of forest biodiversity. Other targets include the four Global Objectives on Forests agreed upon under the United Nations Forum on Forests (UNFF) and the Millennium Development Goals (MDGs). For example, by 2015, the four Global Objectives aim:

- to reverse the loss of forest cover worldwide through sustainable forest management (SFM), including protection, restoration, afforestation and reforestation;
- to enhance forest-based economic, social and environmental benefits and the contribution of forests to the achievement of internationally agreed development goals;
- to increase significantly the area of protected forests worldwide and other areas of sustainably managed forests;
- and to reverse the decline in official development assistance (ODA) for SFM and mobilize significantly increased new and additional financial resources from all sources for the implementation of SFM.

There is a high degree of overlap and complementarity between the draft Strategic Plan of the CBD and these targets, for example, in the aim to increase the area of protected forests worldwide. Similar overlap and complementarity exists between the CBD Draft Strategic Plan and the Millennium Development Goals, in particular goal 7, which aims to ‘ensure environmental sustainability’, and includes sub-target 7b to achieve a significant reduction in the rate of biodiversity loss, with one of the indicators being the proportion of land area covered by forest. At the country level, targets have been set by several countries to achieve this MDG sub-target. Following are several examples: Mongolia aims to increase forest cover from 8.2% to 9.0% from 2000 to 2015; Bhutan, aims to maintain at least 60% of the country under forest cover in perpetuity; and Romania aims to increase afforestation rate from 27% to 35% by 2040 (MDG National Targets).

Further research needs

The achievement of the targets contained in the CBD draft Strategic Plan, and other international

agreements, requires monitoring and scientific analysis of the data collected for indicators. Monitoring for example of the 2010 biodiversity target initially suffered from a lack of data, a lack of methodologies to analyze the data which was available, and a lack of agreement on clear indicators.

Today, despite better data availability on biodiversity and better criteria and indicators to measure it, further research and development needs exist. In relation to forest biodiversity these needs include the following.

- further quantifying of the economic benefits of forest biodiversity to highlight the growing costs of biodiversity loss and ecosystem degradation, as begun for example for the study on The Economics of Ecosystems and Biodiversity (TEEB),
- improving the monitoring of forest biodiversity at the national level, in particular in developing countries, and the development of using easy-to-use yet robust methods to do this,
- refining and/or operationalizing the definitions of certain terms, such as forest degradation and the classification of forest types,
- analyzing the patterns of success or failure in reducing the trend of forest biodiversity loss, both at the national and local level,
- providing indications of the feasibility of certain medium- or long-term political targets, such as the potential for large-scale forest landscape restoration,
- developing maps that overlay carbon storage and biodiversity at different scales,
- improving the understanding of the roles of functional species in ecosystem processes,
- developing meaningful thresholds for the indicators to establish firm targets for Sustainable Forest Management, and
- analyzing opportunities and potential risks of REDD-plus efforts for biodiversity.

Opportunities to provide input in 2010

The CBD holds a series of key events in 2010 which provide unique opportunities for the scientific community to provide policy-relevant information to political decision makers, including on the research needs we just outlined. Scientific input can be given at these events, for example, by disseminating information papers, organizing side events or informing party submissions.

In May 2010, the targets of the CBD draft Strategic Plan will be discussed at meetings of the Convention's Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA) and its Working Group on the Review of its Implementation (WGRI) in Nairobi. SBSTTA and WGRI will forward recommendations to the tenth meeting of the Conference of the Parties to the CBD (COP 10), which will take place in Aichi-Nagoya from 18-29 October 2010. A key stepping stone in the run-up to COP 10 will be a high level session of the

United Nations General Assembly in New York in September which will feature heads of states.

The high level session of the United Nations General Assembly takes place in the context of International Year of Biodiversity that the UN are celebrating this year. The end of the International Year on Biodiversity will be celebrated in a closing ceremony in Kanazawa, Japan, on 18-19 December 2010. The closing ceremony will also provide a seamless bridging to the International Year of Forests which has been proclaimed by the United Nations for 2011.

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The struggle to mitigate negative impacts of invasive alien species in Japan

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Abstract

Japan enacted the Invasive Alien Species Act in 2004 to control nonindigenous species that are recognized as, or suspected of, causing damage to ecosystems, human safety, agriculture, forestry, or fisheries. Under the act, raising, planting, keeping, or transporting invasive alien species (IAS) is prohibited without the express permission of the relevant minister. Difficulties in controlling IAS in Japan stem from the country's reliance on imports, and the act represents a revolutionary advance in biological conservation. In some cases, however, enforcing the act has been difficult. For example, the number of exotic stag beetles that has been introduced and bred as pets is too large to control under the present budget. In the case of the European bumblebee, which was introduced as a pollinator of agricultural crops, designation as an IAS involved a bitter controversy between proponents of biological conservation and agricultural productivity. The act also has a serious loophole in that it does not encompass microorganisms. Thus, how to react to the invasion by amphibian chytridiomycosis has caused confusion among scientists and the Japanese government.

Keywords: Invasive Alien Species Act, Risk assessment, Bumblebee, *Bombus terrestris*, Chytridiomycosis, *Batrachochytrium dendrobatidis*, stag beetle, *Dorcus titanus*.

Introduction

Invasions of alien species began in ancient times, when humans started to migrate between continents. The number of alien species and their migration distances have increased rapidly, since the days of the great 15th-century explorers. Invasive alien species (IAS) are now considered to be one of major causes of species extinction in the world (Millennium Ecosystem Assessment, 2006). Furthermore, IAS are increasingly becoming economic and social problems. Annual economic losses caused by introduced pests to crops, pastures, and forests in the United States, United Kingdom, Australia, South Africa, India, and Brazil amount to nearly US\$230 billion, and annual environmental losses are over \$100 billion (Pimental et al., 2001). Costs of worldwide damage are estimated to be more than \$1.4 trillion per year, representing nearly 5% of the world economy. Assessment and prevention of damage by IAS is of obvious importance.

Control of IAS universally includes (1) risk

assessment prior to introduction; (2) appropriate control measures following introduction; (3) eradication of harmful IAS; and (4) replacement of useful alien species by native species with similar functions. Because populations of IAS can increase and migrate autonomously and can adapt to new environments, controlling their impacts once they are established in the field is extremely difficult. Thus, preventing the introduction of IAS is much more important than controlling IAS that are already established (International Union for Conservation of Nature, 2000). Prevention requires a check and quarantine system to detect and eradicate IAS introduced by transfer of humans and goods. Some alien species that offer high productivity or other economic benefits, such as plants and agricultural biomaterials, may be introduced intentionally. These species should be completely controlled and not allowed to become established in the field. Risk assessment based on scientific data and an effective regulation system are needed.

Once an undesirable IAS is established, the possible responses are to eradicate, control, or disregard it. Because our previous understanding of associated problems was superficial, we disregarded the establishment of many IAS. However, ecological information that is currently available has greatly improved our ability to predict and measure potential impacts of IAS. If any detrimental impacts are predicted or observed, actions to eradicate or control the IAS should be taken. Such actions will be most effective when the establishment area is restricted. If an IAS has already expanded its distribution, the cost of control and the probability of success should be assessed. If control is expected to be ineffective, the IAS should be disregarded. Because resources (money, time, and manpower) are limited, control of various IAS must be prioritized, based on predicted ecological and economic impacts.

Prevention and control of IAS depend on development of legal systems that impose penalties against their introduction and, if already established, IAS may require eradication. The current review discusses the Invasive Alien Species Act, which was enacted in Japan to control IAS. Several controversial case studies are presented, including introduction of exotic stag beetles and the European bumblebee. Finally, loopholes in the law, related to the introduction of microorganisms, will be examined, with specific reference to chytridiomycosis.

Japan's Invasive Alien Species Act

Japan depends heavily on international trade, importing large volumes of goods, including living organisms. Many unwanted species are unintentionally brought into the country with the imported goods and in the transport containers. The present quarantine system was established within the framework of the International Plant Protection Convention and the World Organisation for Animal Health. It was designed to prevent adverse effects of IAS on agriculture, forestry, and fisheries, but does not protect wild fauna and flora or ecosystems.

The Japanese government created the Invasive Alien Species Act in 2004 to implement the provisions of Article 8(h) of the Convention on Biological Diversity (1992) and to comply with COP 6 decision VI/23, which was adopted at the Sixth Ordinary Meeting of the Conference of the Parties to the Convention on Biological Diversity, in April 2002. The act came into force in July 2005. Its purpose is to control IAS and to prevent damage to ecosystems caused by IAS.

The Invasive Alien Species Act defines IAS as alien species recognized as, or suspected of, causing damage to ecosystems, human safety, agriculture, forestry, or fisheries (Fig. 1; www.env.go.jp/en/nature/as.html has the full text of the act). Species considered alien are limited to those introduced into Japan since the Meiji era (ca. 1868), when Japan's trade with the rest of the world and the introduction of alien species markedly increased. Species are assessed on whether they will predate native species, compete with native species for ecological niches, disturb reproduction of native species by interspecies crosses, or disrupt native ecosystems. Relevant ministers have the responsibility of assessing and deciding, on advice from scientific experts, which species should be designated as IAS. IAS are subject to various regulations: raising, planting, keeping, or transporting them is prohibited without the express permission of the relevant ministers. Permission is a prerequisite for importing IAS, and releasing them into the wild is not allowed at any time.

The act additionally defines species belonging to the same genus or family as IAS as uncategorized alien species (UAS), based on the possibility of similar ecological impacts. For example, the Taiwan macaque (*Macaca cyclopis*) competes with the Japanese native macaque *M. fuscata* and is designated as an IAS; therefore, most other species in the genus *Macaca* are designated as UAS. Before importing UAS into Japan, importers must notify the relevant ministers and provide information on the ecological properties of the UAS. The species are evaluated by experts within six months of the application. UAS determined to pose a risk are immediately designated as IAS, while those posing no risk are permitted. In

2008, six reptile and six amphibian UAS were evaluated and judged to be IAS.

Although the Invasive Alien Species Act imposes controls on the importation of designated species, it would be more effective to prohibit nearly all alien species. This is the case in Australia and New Zealand, where importation of all alien species is prohibited, except those designated as safe. This is considered a "white-list system," while the Japanese system is a "black-list system." Although Japan would benefit from adopting a white-list system, the country's economic situation makes this approach difficult. Most of Japan's natural resources are imported from other countries, and a white-list system could impose obstacles to natural resource supply.

Designation of IAS

The Japanese government initiated the designation of IAS in 2004. A general expert meeting and six working groups were established to evaluate each group of alien species (mammals and birds, reptiles and amphibians, fish, insects, other invertebrates, and plants). Two special working groups discussed the largemouth bass (*Micropterus salmoides*) and the European bumblebee (*Bombus terrestris*). Largemouth bass were illegally released into lakes and ponds throughout Japan for game fishing and subsequently caused damage to fisheries and native fish species. The European bumblebee has been widely used as pollinators in Japanese greenhouses. The sports fishing and farming industries were concerned about the designation of those species as IAS and the possible prohibition of their use.

Each working groups identified species that were already reported by international scientific publications to have adverse effects and made recommendations to the general expert meeting, which decided on the final recommendations to the ministers. After public consultation, the ministers finalized the decision. As a result, about 100 species were designated as IAS (www.env.go.jp/nature/intro/1outline/files/siteisyu_list_e.pdf). More than 110,000 comments were received from the general public; most were against designation of the largemouth bass as an IAS. Newspapers and other media reported the conflict, resulting in a rapid increase in public awareness of the Invasive Alien Species Act.

Alien species that were already widely established in Japan prior to enactment of the Invasive Alien Species Act, such as the red swamp crawfish (*Procambarus clarkii*), and red-eared slider (*Trachemys scripta elegans*), have never been listed as IAS, due to difficulty of control. IAS designation of African lovegrass (*Eragrostis curvula*), which impacts riverbed flora (Matsumoto et al., 2000), has been delayed, because the species is often planted at construction sites, and no alternative species have been found. This situation reflects the precedence of

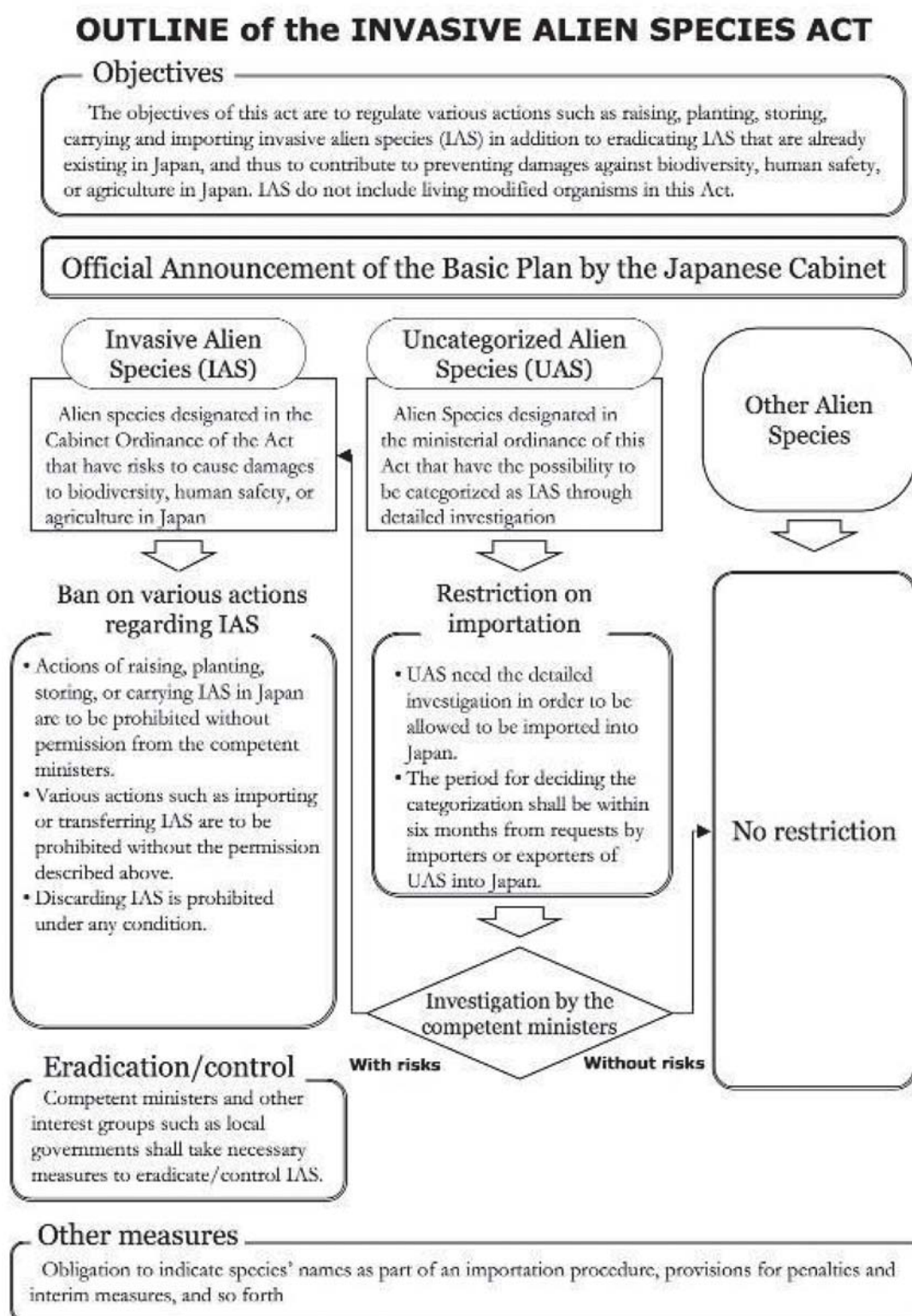


Fig.1 Outline of Japan's Invasive Alien Species Act.

economy over environmental health in Japan.

Three species provide particularly useful examples of the struggle to mitigate negative impacts of IAS in Japan: the European bumblebee; exotic stag beetles, such as *Dorcus titanus*; and the chytrid fungus *Batrachochytrium dendrobatidis*. The case history of each of these species is presented in detail.

European bumblebee, *Bombus terrestris*

1) Controversy over an alien pollinator

The European bumblebee is one of the most successful biological agents used for commercial pollination. The industry breeding this species has flourished worldwide since the 1980s and has helped to increase rates of agricultural productivity (Ruijter, 1996). The European bumblebee was introduced to Japan in 1991, primarily for pollination of tomatoes. By 2004, the number of commercial colonies used annually reached almost 70,000 (Kunitake and Goka, 2006). Use of *B. terrestris* not only increases tomato crop productivity, but also reduces use of chemical pesticides that could weaken the bee's activities, ultimately increasing the quality and safety of tomato products. However, many ecologists and entomologists have warned of the ecological risks posed by *B. terrestris* (Goka, 1998, 2003; Washitani, 1998). A naturalized colony was found in Hokkaido in 1996 (Washitani, 1998). Since then, the number of captive colonies escaping into the field has continued to increase, suggesting that the rate of invasion by the alien bee is increasing (Matsumura et al., 2004). Many scientists fear that the alien bee will eliminate native species of bumblebee through competition, based on the similarity of their ecological niches (Washitani, 1998), and have argued that *B. terrestris* should be designated as an IAS. At the same time, agriculturalists concerned about conserving productivity have objected to legal regulation of *B. terrestris*. Thus, the alien bee has created a controversy between proponents of biological conservation and agricultural productivity.

Although conservation of the Japanese ecosystem is the first policy of the law, another of the law's policies is that the socioeconomic background of the use of introduced species should be considered in full before decisions are made to regulate a species. Therefore, because the introduction of *B. terrestris* had both an economic aspect through its contribution to agricultural productivity and a social aspect through improvement of the living standards of farming families because of the improved productivity, caution was required in making a decision to declare *B. terrestris* an IAS.

2) Impact and risk assessment of *B. terrestris* as an alien pollinator

In light of the above situation, the Japanese Ministry of the Environment set up a Bumblebee Specialist Group to discuss management of *B. terrestris*. The

group listed four ecological risks posed by *B. terrestris*: (1) exclusion of native pollinators, through competition for food and nest sites; (2) inhibition of reproduction by native plants, through disturbance of natural pollination in ecosystems; (3) disturbance of reproduction by native bumblebees, through interspecies crosses; and (4) introduction of alien parasites that could be pathogenic to native species. In 2005, the National Institute for Environmental Studies initiated a study, "Development of control methods for ecological risks posed by introduced bumblebees," supported by the Research Project for Utilizing Advanced Technologies in Agriculture, Forestry, and Fisheries, in collaboration with other institutes, universities, private companies, and the government. The results of the study confirmed the four ecological impacts (Goka et al., 2006, 2007; Dohzono et al., 2008; Inoue et al., 2008; Kondo et al., 2009; Inoue and Yokoyama, 2010). Consequently, *B. terrestris* was declared to be an IAS, an invasive threat to Japanese native fauna and flora, at the Specialist Group meeting held in December 2005, and regulation according to the Invasive Alien Species Act was required.

3) Controlled use of *B. terrestris* as an alien pollinator

In addition to evaluating the impact of *B. terrestris* on native species, the Bumblebee Specialist Group considered permitting use of the species for agriculture in secure facilities that would prevent its escape. As part of the study, "Development of control methods for ecological risks posed by introduced bumblebees," a technique for covering greenhouses with nets that completely prevent the escape of the bumblebees from the greenhouses (Fig. 2) was developed (Koide et al., 2008). A statistical method for estimating the number of *B. terrestris* nests naturalized in the field was also developed to monitor escape of the bees (Kokuvo et al., 2007, 2008). On the basis of these control and monitoring methods, the Ministry of the Environment adopted a permission system for use of the alien pollinator. Since March 2007, farmers have been required to obtain permission from the Ministry of the Environment before using *B.*

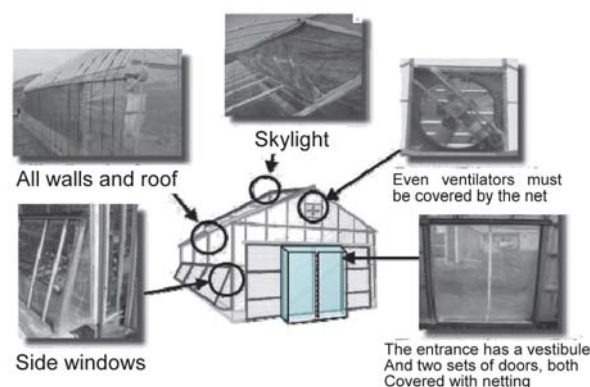


Fig. 2. The use of netting to prevent escape of *Bombus terrestris* from greenhouses (modified from Koide et al., 2008).

terrestris and to completely cover their greenhouses with nets.

The decision to legally regulate *B. terrestris* in Japan was momentous from two perspectives. First, it demonstrated that even beneficial species, such as pollinators, can be regulated by law, if the species are designated as IAS. Second, it demonstrated that the law can resolve controversies between conservation ecology with agricultural productivity. Nevertheless, many obstacles must be overcome for this test of the Invasive Alien Species Act to succeed. Farmers must be educated on the need and methods to control *B. terrestris*, surveillance systems for inappropriate use of *B. terrestris* must be put in place, and farmers need help in covering the costs of controlling *B. terrestris*.

4) Alternatives to the controlled use of *B. terrestris* as a pollinator

The Ministry of Agriculture, Forestry and Fisheries has recommended use of the native bumblebee *Bombus ignitus* as an alternative pollinator of tomato plants. This species is not regulated, and farmers cannot be penalized if it escapes. However, even a native species poses a risk of acting as an IAS when it is artificially transported beyond its natural habitats. For example, crosses between the natural and commercial colonies could cause genetic introgression. DNA analysis has shown genetic diversity among local populations of *B. ignitus*, indicating the need to examine genetic endemism before commercialized colonies are used (Tokoro et al., 2010). Recent shortages of the European honeybee (*Apis mellifera*) all over Japan have increased the demand for bumblebees as alternative pollinators for a variety of agricultural crops. Diversified use of *B. ignitus* could make the ecological impacts of commercial colonies more difficult to assess and control.

Exotic stag beetles

1) Commercialization of stag beetles in Japan

The breeding of stag beetles as pet animals has become very popular in Japan since 1999, and many exotic species have been imported from other countries (Goka et al., 2004). Over one million exotic stag beetles are currently imported into Japan annually, some via criminal smuggling. The ecological risks of this unprecedented insect trade are not well understood. Native stag beetles will probably suffer serious and direct impacts from escaped pets, due to them having similar ecological niches (e.g., through competition for food and habitat). The exotic beetles may carry parasitic invaders into Japan. Finally, hybridization may result in genetic introgression between the exotic and native populations.

2) The risk of genetic introgression caused by exotic stag beetles

Dorcus titanus, one of most popular species of stag beetles in Japan, is widely distributed among the

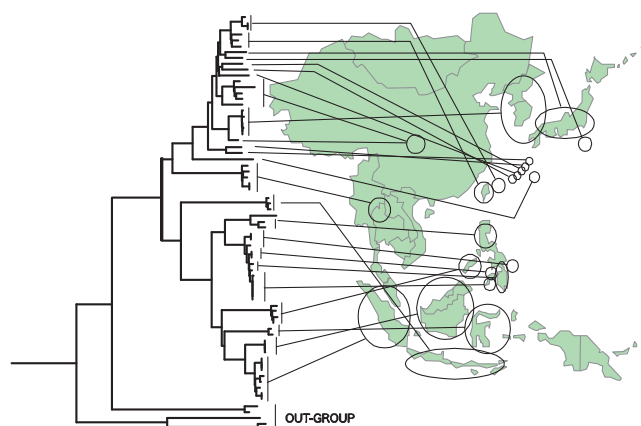


Fig.3 Phylogenetic relationships among Asian populations of the stag beetle, *Dorcus titanus*, based on the mtDNA-CO gene (new data added to the phylogenetic tree in Goka et al., 2004).

Japanese Islands. Subspecies of *D. titanus* and closely related species are widely distributed throughout Asia and the islands of Southeast Asia. Many individuals of these exotic subspecies have been imported to Japan for commercialization. To establish conservation units, variation in mtDNA sequences among populations was studied. Molecular genetic analysis indicated that populations from Japan and other geographic areas constitute different phylogenetic lineages (Fig. 3), representing more than a million years of evolutionary history (Goka et al., 2004). Each clade in the phylogram is an evolutionarily significant unit (ESU), according to the definition of Ryder (1986). These unique ESUs could easily be disrupted by genetic introgression as a consequence of hybridization among different populations. Therefore, genetic diversity among populations of *D. titanus* should be carefully considered in formulating policies that regulate transport of the beetles in the pet trade.

To test the ability of geographically distant populations of *D. titanus* to hybridize, exotic beetles, which differ in body size and mandible morphology, were crossed with native Japanese beetles in the laboratory. Almost all hybrid males, which possessed large bodies and intermediate mandible morphology, were fertile. Thus, there appears to be little reproductive isolation, and genetic introgression is likely to occur as a consequence of naturalization of exotic strains in Japan. In fact, a few individuals that possessed mtDNA from exotic populations have already been collected in Japan.

3) Educating the public about the problem of introducing exotic stag beetles as an IAS

The Japanese Ministry of the Environment held a meeting of specialists to discuss how the Invasive Alien Species Act should deal with exotic stag beetles. The number of exotic stag beetles bred in Japan was estimated at half a billion, and many specialists considered control impossible if the beetles are released into the field before they are banned.

Therefore, rather than designating the exotic stag beetles as IAS, the Ministry of the Environment designated them as “Alien Species to Notice,” which can be brought into the country and bred, but should not to be released into the field. The ministry and collaborating scientists have now developed programs to educate the public about the introduction of exotic stag beetles as an IAS problem.

The chytrid fungus *Batrachochytrium dendrobatidis*

1) Invasion of a microorganism that causes an infectious disease

There is a large loophole in the Invasive Alien Species Act: It does not encompass alien microorganisms. Under the present version of the law, only alien species that can be visually identified can be regulated; species too small to see, such as viruses, bacteria, and fungi, are beyond the scope of the law. Although Japan has laws designed to control infectious diseases and parasites that affect humans and domestic animals and plants, these laws do not cover wildlife. Infectious diseases that affect wildlife populations are currently emerging at unusually high rates and pose a serious threat to the conservation of global biodiversity (Harvell et al., 1999; Ward & Lafferty, 2004; Lebarbenchon et al., 2008). Given that more than 500 million live animals are imported into Japan annually (www.forth.go.jp/mhlw/animal/page_b/b03-8.html), the microorganisms that accompany these animals are a cause for concern.

Japan is now facing invasion by chytridiomycosis, a serious worldwide disease of amphibians, caused by *B. dendrobatidis*. The fungus was first identified by Berger et al. (1998) and described by Longcore et al. (1999). Chytridiomycosis is blamed for declines in wild frog populations in Australia, New Zealand, the United States, Central America, South America, and Spain (Berger et al., 1998; Lips, 1999; Pessier et al., 1999; Bosch et al., 2001; Bradley et al., 2002; Green et al., 2002; Ron et al., 2003; Weldon et al., 2004). Japan imports amphibians as scientific and medical materials, pets, and food for pet reptiles and aquarium fishes. These imports come from many countries, including those in Central and South America (Japan Wildlife Research Center, 2008), which are hotspots of *B. dendrobatidis* outbreaks. Furthermore, the African clawed frog (*Xenopus laevis*) and the American bullfrog (*Rana catesbeiana* = *Lithobates catesbeianus*) are alien species that have already become naturalized in Japan (Maeda & Matsui, 1989; Arao & Kitano, 2006). *Batrachochytrium dendrobatidis* globally infects introduced populations of both species (Weldon et al., 2004; Garner et al., 2006).

The amphibian chytrid fungus had never been reported in Asia prior to December 2006, when it was found on a pet frog imported to Japan from South

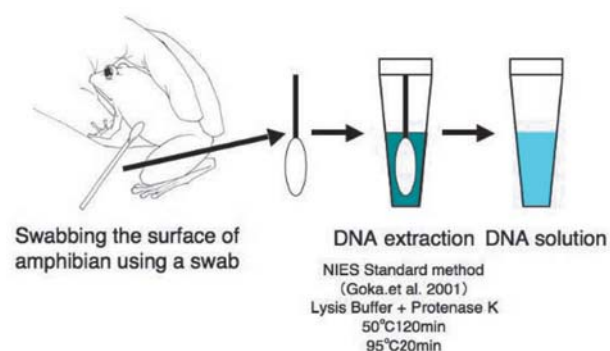


Fig. 4. The use of swab samples to collect *Batrachochytrium dendrobatidis* DNA from amphibians.

America (Une et al., 2009). Many Japanese herpetologists and ecologists panicked when the disease was found, and the media reported the news as a crisis for Japanese amphibians. The current situation was considered as a serious pandemic threat by *B. dendrobatidis* within Japan. To protect native Japanese frog populations against this disease, the current status of infection among captive and free-ranging frogs must be determined, as well as the virulence of the fungus when it infects native species.

2) A rapid-response surveillance system for *B. dendrobatidis* in Japan

A swab sampling method was used to identify amphibian chytrid fungal infections: the surface of each amphibian was swabbed, and DNA was extracted from the swab samples (Fig. 4). Because the samples contained many contaminants, a nested polymerase chain reaction (PCR) assay was developed to obtain specific and highly concentrated products of the fungal internal transcribed spacer (ITS) gene (Goka et al., 2009). A system for inspecting amphibians was established in collaboration with universities, nongovernmental organizations, the Ministry of the Environment, local governments, breeders, and veterinarians (Fig. 5). Swab samples were collected from 265 amphibians sold at pet shops, 294 bred at institutes, and more than 5000 in the field from northern to southwestern Japan (Goka et al., 2009; Goka et al., in preparation).

Results of nested PCR assays of the samples showed *B. dendrobatidis* infections in native and exotic amphibian species, both in captivity and in the field (Goka et al., 2009). Sequencing of the PCR products revealed that native Japanese amphibians carry more than 30 haplotypes of the *B. dendrobatidis* ITS region, including the haplotype reported worldwide (A type, accession no. AY997031). Phylogenetic analysis of these haplotypes, combined with 48 ITS-DNA sequences previously detected in other countries (United States: Geartner et al., unpublished; Rodriguez et al., 2009; Ecuador: Geartner et al., unpublished; Italy: Federici et al., 2009) and included in the DDBJ/EMBL/GenBank International DNA Database, showed that genetic

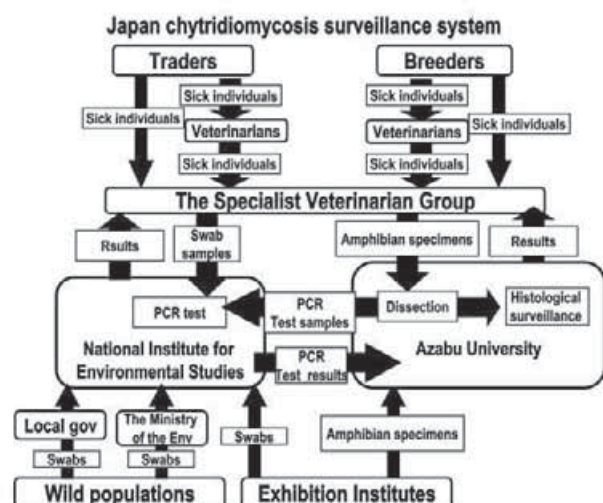


Fig. 5 The rapid-response surveillance system constructed to identify amphibian chytrid fungal infections in Japan.

diversity of *B. dendrobatidis* was higher in Japan than in other countries (Fig. 6; Goka et al., 2009).

Three of the *B. dendrobatidis* haplotypes detected in Japan (B, J, and K) appear to be specific to the Japanese giant salamander (*Andrias japonicus*) and seem to have established a commensal relationship with this native amphibian. Although the incidence of infection in the giant salamander was high (>40%), no disease symptoms were detected (Goka et al., 2009). The highest *B. dendrobatidis* infection rate (>60%) and genetic diversity were found in the sword-tail newt (*Cynops ensicauda popei*) which is endemic to Okinawa Island. The American bullfrog showed the next highest genetic diversity of *B. dendrobatidis* and also relatively high incidence of infection (~20%). Another alien species, the African clawed frog, which is believed to be the original host of the chytrid fungus,

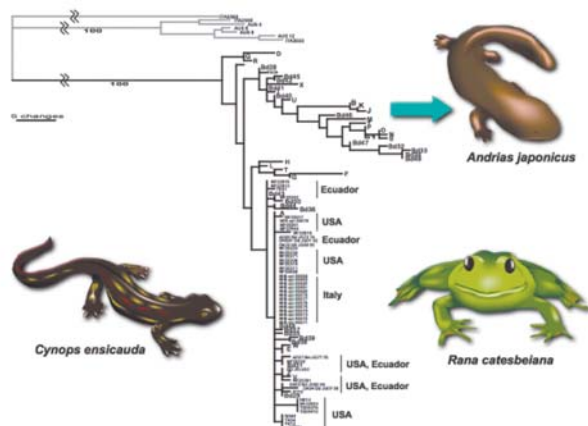


Fig. 6. Phylogenetic relationships among ITS haplotypes of *Batrachochytrium dendrobatidis*, based on maximum parsimony analysis. Gray branches indicate outgroups. Large, boldface letters indicate haplotypes of *B. dendrobatidis* detected in Japan. Small, regular letters indicate haplotypes detected in the United States, Ecuador, and Italy (new data added to the phylogenetic tree in Goka et al., 2009).

showed a high incidence of infection, but relatively low fungal genetic variation (Goka et al., 2009; Goka et al., in preparation). Except for the high incidence of infection in *C. ensicauda popei* and *A. japonicus*, the overall incidence of *B. dendrobatidis* infection in Japanese native amphibians was less than 1%.

Rana catesbeiana has been proposed as one of the key species responsible for expanding the global distribution of *B. dendrobatidis* by acting as a carrier of the fungus (Daszak et al., 2004; Ouellet et al., 2005; Garner et al., 2006; Fisher & Garner, 2007). However, we found no evidence that the incidence of infected amphibians increased around habitats where infected individuals of *R. catesbeiana* were detected. On the contrary, sites of infected native species were often not adjacent to sites of infected *R. catesbeiana*. For example, *C. ensicauda popei* in a natural forest on Okinawa Island, where no alien amphibians were found, had a relatively high incidence of *B. dendrobatidis* infection (Goka et al., 2009). These results suggest that horizontal infection between *R. catesbeiana* and native species is not the primary source of infection.

The low incidence of *B. dendrobatidis* in most native amphibian species, and the fact that no disease symptoms have been reported in native amphibians in captivity or in the field, suggest that the fungus is endemic to Japan, and that many native species can tolerate infection by the fungus. The high genetic diversity and endemism of *B. dendrobatidis* in Japan may provide support for a new hypothesis, namely that the fungus originated in Japan or some other part of Asia. If so, the various haplotypes of chytrid fungus found on alien amphibian species in Japan might have been propagated from cryptically infected amphibians native to Japan.

If *B. dendrobatidis* originated in Japan or other parts of Asia, trade in amphibians as food resources or pet animals between Asian countries and other geographic areas might have enabled the fungus to expand its distribution worldwide. In that case, spread of the fungus should have been primarily in urban areas and other sites where amphibians are artificially transported. However, damage to wild amphibians in Latin America and Australia has occurred mainly in tropical highland rainforests, which are relatively unexplored and undeveloped. Recent development, alteration of landscapes, and expanding eco-tourism might account for the spread of *B. dendrobatidis* in such areas.

3) Implications of *B. dendrobatidis* in Japan

The genetic diversity and endemic nature of *B. dendrobatidis* in Japan suggests cospeciation of the fungus and amphibian hosts (i.e., each fungal strain is specific to a natural host). Anthropogenic disturbances of the environment and artificial transportation of amphibians undoubtedly carry the chytrid fungus from its native habitats into nonnative habitats. Under these conditions, the alien fungus must switch to a new host

amphibian if it is to survive, and unnatural combinations of amphibians and fungal strains that have not coevolved may explain the resulting pandemic. This scenario fits the model of emerging diseases that threaten human health, such as AIDS and SARS (Daszak et al., 2000; Lebarbenchon et al., 2008). Thus, the case of *B. dendrobatidis* confirms the significance of biodiversity conservation from the viewpoint of epidemiology.

Now that the situation is better understood, the argument that the introduction of chytridiomycosis would create a crisis for Japanese amphibians can be seen as presumptive. It is clear that we need firm scientific data to predict the risks associated with each IAS, and that we need to construct and maintain monitoring and research systems to support risk assessment. Without a systematic, scientific approach, we risk overlooking real threats posed by IAS.

Conclusion

The three cases presented in this paper demonstrate that the first response to IAS should be rapid accumulation of scientific data. Although the European bumblebee is a useful alien species, scientific data provided proof of their invasiveness. Careful research on DNA variation and hybridization showed that commercial introduction of exotic stag beetles could destroy native populations with evolutionary significance. Genetic assays of more than 5000 amphibians revealed a possible and unexpected origin of chytridiomycosis.

The number of IAS in Japan continues to increase, even after passage of the Invasive Alien Species Act. The ecology and ecological impacts of IAS are and will continue to be extremely diverse. Unnatural combinations of native species and nonnative habitats will undoubtedly have unexpected impacts, and the ecological risks need to be analyzed from many aspects. To mitigate the impacts of IAS, the public must understand the precarious situation caused by Japan's strong economic dependence on international trade (Goka et al., 2004).

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