# SATELLITE IMAGES AS PRIMERS TO TARGET PRIORITY AREAS FOR FIELD SURVEYS OF INDICATORS OF ECOLOGICAL SUSTAINABILITY IN TROPICAL FORESTS

by

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**To my aitas Amatxu & Pepe Mandy**

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University of Nebraska, 2006

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Sustainable management of tropical forests has been identified as one of the main objectives for global conservation of carbon stocks. In order to achieve this, managers need tools to establish whether or not their management practices are sustainable. Several tool development initiatives have undertaken the creation of sets of criteria and indicators to aid managers to target, if not achieve, sustainability. The question of how to assess these indicators remains to be answered from an operational viewpoint, where logistical constraints become critical and priorization becomes necessary. The present dissertation sought to determine whether satellite imagery can be used, in conjunction with standard forest management data, to identify priority areas for field surveys of indicators of ecological sustainability of managed tropical forests. It presents a novel approach to the assessment of CIFOR indicator I.2.1.2: *"The change in diversity of habitats as a result of human interventions is maintained within critical limits as defined by natural variation and/or regional conservation objectives"* by means of semivariography of remote sensing data. It shows the Wide Dynamic Range Vegetation Index (WDRVI) is a good alternative for the detection and quantification of tropical forests structural heterogeneity and its dynamic change. The differences observed between forest management units and natural areas forest structural heterogeneity were used to identify priority areas for field survey of ecological sustainability indicators and evaluate how these priorities were reflected in dung beetles community structure and composition. The link between forest structural heterogeneity dynamic change, forest logging intensity and dung beetle community structure and composition is established. A logging intensity threshold of 4 trees per hectare is identified as the limit between significant or not significant differences in forest structure dynamic changes and dung beetles community total species richness and diversity estimates.

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# Introduction

Tropical forests have been identified as the largest and most endangered of biomes. Forests which are neither economically productive nor protected by conservation status are at high risk of transformation into other land uses that are considered more productive, such as agriculture. Sustainable management constitutes then, a reasonable alternative from economic and conservation perspectives. As not all management practices are sustainable, several international initiatives have undertaken the development of criteria[1] and indicators[2] (C&I processes) to be used by managers as tools for directing their practices towards sustainability within a given timeframe and with the ability to monitor trends (Wijewardana 1998; CIFOR 2000). These initiatives include a wide variety of indicators that must be assessed to build an integrated view of sustainability. The majority of the ecological sustainability indicators are based on a scientific or heuristic approach (Stork *et al.* 1997). How best to implement these indicators remains unanswered from an operational standpoint (McGinley and Finegan 2003).

In 1994, the Center for International Forestry Research (CIFOR) assembled expert teams to address how the ecology, economics, and production aspects of forestry affect management sustainability and to design specific sets of C&I based on research, conceptual frameworks, and field evaluations. The process resulted in a generic template that forest managers can use and adapt to the specific conditions of their operations (CIFOR 2000). They presented biodiversity maintenance as a surrogate for ecological sustainability and, thus, most of the ecological sustainability indicators are oriented towards assessing the various dimensions of biodiversity across taxa.

Dung beetle biodiversity and composition has been shown to relate to forest structural heterogeneity and habitat diversity (Halfter and Favila 1993), which is also related to the heterogeneity of forest canopy reflectance (Tuomisto et al. 1995, 2003; Tuomisto 2003). In 1999 as part of my M.Sc. research (Aguilar-Amuchastegui 1999) at the Tropical Agronomic Research and Higher Education Center (CATIE) in Costa Rica, I evaluated the utility of dung beetle

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[1] A standard that a management (any aspect) is judged by (CIFOR 2000)

[2] Any variable or component of the forest ecosystem or management system used to infer the status of a particular criterion.(CIFOR 2000)

(Coleoptera: Scarabaeinae) biodiversity for forest workers and managers to address three important CIFOR indicators; specifically, I.2.1.4. *The richness/diversity of selected groups shows no significant change*, V.2.1.4.1 *Species richness of prominent groups is maintained or enhanced*, and V.2.1.4.6. *Temporal change in species richness is not significant* (CIFOR 2000). Dung beetles have been repeatedly proposed as sensitive indicators of biodiversity due to their intimate relationships with all types of vertebrate faunal dung and as key decomposers on the forest floor. Further, they play important roles in forest regeneration, such as seed dispersal and nutrient cycling (Cambefort and Hanski 1991; Halfter and Favila 1993; Favila and Halftter 1997; Aguilar-Amuchastegui 1999, Davis *et al* 2001). My research was conducted in Costa Rica, Central America. Dung beetles were surveyed again in 2000 and 2001.

I found that dung beetle biodiversity provided a viable assessment tool for ecological sustainability in conjunction with standard forest stand inventory data (Aguilar-Amuchastegui et al. 2000). Forestry workers were able to conduct field surveys after only basic training in beetle taxonomy and managers (mostly forest engineers) were able to interpret these data to make management decisions to maintain forest productivity and carbon stocks. Both dung beetle community diversity and composition and the assessment methods used are now proposed by the World Wildlife Fund (WWF) and CATIE (Finegan *et al.* 2004) for use in other types of monitoring.

However, as management areas increase in size and number, logistical constraints facing field surveys become critical and a prioritzatiopn process becomes necessary. Most CIFOR (2000) indicators and verifiers are usually surveyed in the field. Spatial coverage of the field surveys is necessarily restricted and even then requires significant skilled labor (Tuomisto et al. 1995, 2003). The problem facing all tropical forest managers is how to assess these bioindicators effectively across large management areas without exhaustive surveys that would be prohibitively expensive and time consuming? A prioritizing approach is clearly needed. Yet there are currently no generic approaches for constructing it.

Tropical forests are not steady-state ecosystems; they exhibit a range of natural disturbance levels (Finegan 1996). Their historical range of variability (cf. Morgan *et al.* 1994) embraces a dynamic of disturbances that creates a mosaic of successional stages differing in microclimate, vegetation structure, and floral and faunal composition. The distribution of these stages generates the vertical and horizontal heterogeneity of a forest. This heterogeneity can be changed by forest management (Finegan *et al.* 2004, Morgan *et al.* 1994, Ghazoul and Hellier 2000). Accordingly, CIFOR (2000) established this indicator (I.2.1.2):*The change in diversity of habitats as a result of human interventions are maintained within critical limits as defined by natural variation and/or regional conservation objectives*. In other words, if a management scheme maintains the relative spatio-temporal abundance and distribution of these successional stages within the limits of background variability, it is evidence that the management may be sustainable and the contrary would constitute evidence of potentially unsustainable management.

Forest heterogeneity may be detected remotely by means of spaceborne sensors (Foody and Cutler 2003). Tuomisto et al. (2003) used Landsat TM imagery to identify sites for field surveys of understory plant bioindicators (Melastomataceae, Pterofitidae), based on the characterization of the heterogeneity of reflectance from Amazonian forest canopies. They were able to establish relationships between forest reflectance textural differences, soil patterns, and bioindicator distribution and biodiversity. Working in forested basins in California, Seto et al. (2004) found positive correlations between the diversity of two other groups commonly proposed as biodiversity indicators—birds and butterflies—and the mean, maximum, and variance of TM derived NDVI.

In the present dissertation I investigated whether spatial analysis of remotely-sensed imagery can be used in conjunction with standard forest management data to identify priority areas for field surveys of a widely accepted sustainability bioindicator—dung beetle species diversity and abundance. The aim was to improve the targeting and affordability of ecological sustainability surveys in tropical forests. I established relationships between significant observed changes in forest canopy reflectance heterogeneity, forest management practices, and dung beetle species community composition.

The **primary objective** of the research was to investigate and characterize the relationships between the spatial heterogeneity of canopy reflectance heterogeneity and dung beetle community diversity and composition. The **central hypothesis** was that the spatial heterogeneity of canopy reflectance in tropical forests is related to the spatial heterogeneity of forest successional stages, which is related to vertebrate faunal diversity and composition, which is, in turn, necessarily related to dung beetle community structure and composition due to well-documented specializations of dung beetle species with the particular dung producers (Figure 1).

The **rationale** was that once the relationships between canopy spatial heterogeneity and dung beetle biodiversity were characterized, remote sensing of forest structure may serve as an important practical tool for guiding sustainable management of tropical forests and conserving vulnerable carbon stocks.

1. Quantify the spatial heterogeneity of forest structure.

Research hypothesis:

*More [less] intensely managed forest stands will exhibit [no] significant differences in the spatial heterogeneity of canopy reflectance as observed by spaceborne sensors in comparison with undisturbed reference stands.*

2. Establish relationships between forest structural heterogeneity and dung beetle community structure and composition and model dung beetle diversity as a function of canopy reflectance heterogeneity.

Research hypothesis:

*Managed forest stands that do [not] exhibit significant differences in the spatial heterogeneity of canopy reflectance will also show [no] significant differences in dung beetle species diversity and community composition.*

The long-range goal of the present research is to develop, test, and disseminate a methodology that enables tropical forest managers to use the potential relationship between the spatial heterogeneity of canopy reflectance and dung beetle community diversity and composition as a means of programming field surveys of dung beetles and biodiversity indicators in general. It is my expectation that my results are significant enough to provide the basis for the development of a practical way of using information derived from spaceborne sensors for tropical forests management sustainability assessment (apart from forest classification or inventorying). Such a tool should be usable by any forest management operation and even linked to national policies decision making and surveying, anywhere in the tropics.

### *Dissertation strucrure: how to read this document*

In order to achieve its goal, this disertation embraces current knowledge in tropical rainforest ecology, dung beetle ecology, remote sensing of forest structure, spatial statistics, model selection theory and forest management. This means that most likely, the average reader will have some background in two or three of these areas but will lack of it in the remaining ones. Based on this, this dissertation has been divided in two major sections. The first section (Section I) is constituted by chapters 1 to 4. It seeks to provide the reader with the required basic background in those areas in which he lacks of it. Each chapter was conceived as an independent unit. Doing so enables the reader to cover any knowledge gap he may have, without having to go through the entire document including those chapters pertaining to topics he already manages. The second section (Section II): the “Core” of this dissertation, includes chapters 5: The Study Area, chapters 6 to 8, which correspond to the resulting papers that were written and submitted for publication, organized in a coherent order that allows the reader to go through all the aspects touched by this research, and finally, chapters 9 and 10, which correspond to the conclusions and recommendations. As an initial approach, any given reader may, if he chooses so, read this section first and, in case of need,` because he found out he lacks of some background in one of the related topics, he can refer himself to the specific chapter pertaining the topic in question.

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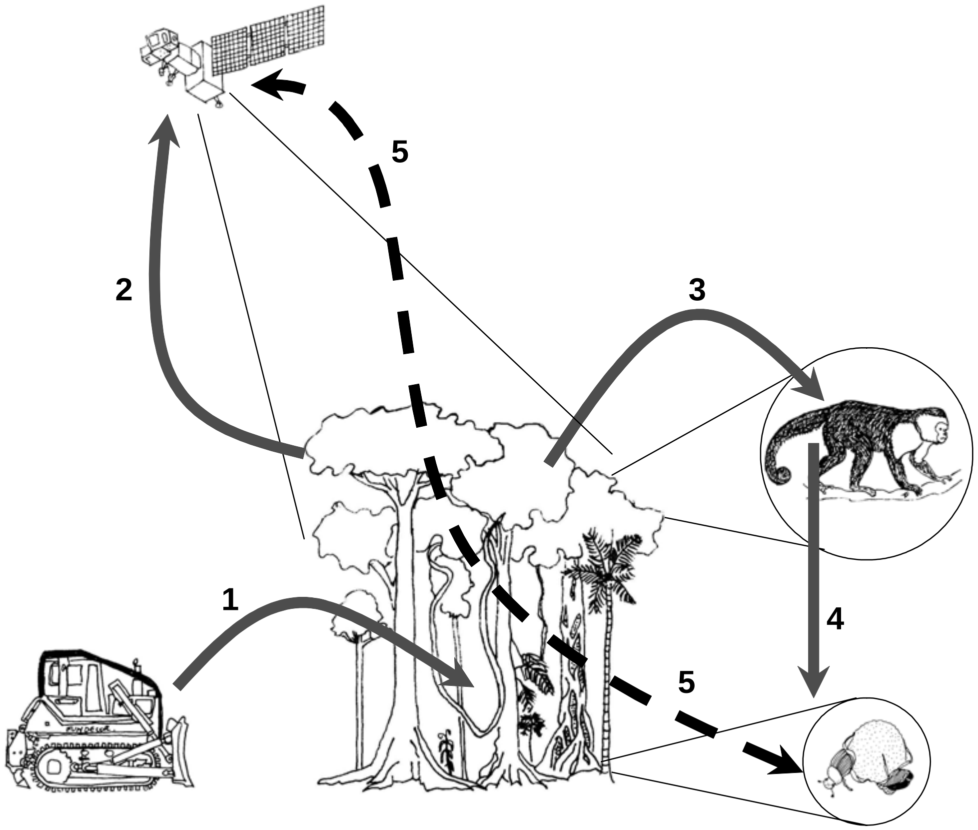

**Figure 1.** Illustrated hypothetical linkages from dung beetle to remote sensor: (**1**) Forest management can change forest structural heterogeneity; (**2**) Forest structural heterogeneity and its change can be measured by remote sensing using semivariography of vegetation indices; (**3**) Forest structural heterogeneity influences habitat quantity and quality for forest vertebrate fauna; (**4**) Forest vertebrate fauna composition influences dung production and thus dung beetle community structure and composition; and (**5**) my central hypothesis: *Managed forest areas that [do not] exhibit significant differences in forest structural heterogeneity as compared with natural areas will also [not] show significant differences in dung beetle community species diversity and composition*.

# Section I

## Chapter 1: The quest for sustainable forest management (SFM)

*Sustainable Forest Management*

Tropical forests have been identified as the largest and most endangered of biomes (Vieira et al. 2004). Even though they cover less than 7% of the earth's surface, they represent 47% (~1830 million ha) of total global forested area (FAO 2005). They house half of the animal, plant, fungal and microbal species in the world (Myers 1992). Tropical forests regulate global climate (Gedney and Valdes 2000), act as carbon sinks (Grace et al. 1995) and provide valuable ecosystem services and resources such as hydrological balance, sedimentation reduction, disaster prevention and biodiversity conservation (Laurance 1999; Kalácaska et al. 2004a,b). Forest areas that are neither protected by a conservation status nor considered as economically productive are at risk of transformation into other land uses that may be considered productive, such as agriculture or pasture. Only a small portion of these forests are under a conservation status (Clarke et al. 2005).

Sustainable forest management (SFM) has been identified as a reasonable alternative to transformation, from economic and conservation perspectives (Greiser Johns 1997; Pokorny and Adams 2003; Hamer et al. 2003; Clarke et al. 2005; Ingram et al 2005). In 1992, during the United Nations Conference on Environment and Development (UNCED), commonly known as The Earth Summit, held in Rio de Janeiro, SFM was identified as one of the key aspects to be considered when assessing sustainable development. In 2004, in the seventh meeting of the Conference of the Parties of the Convention on Biological Diversity (CBD), SFM was considered as one of a series of indicators to be developed and used to report on the previously agreed target for CBD member states: to "achieve by 2010 a significant reduction of the current rate of biodiversity loss at the global, regional and national levels as a contribution to poverty alleviation and to the benefit of all life on Earth" (Dudley et al. 2005).

Even though SMF concept is pretty much well established, no clear definition exists. The International Tropical Timber Organization (ITTO) defines SFM as "*the process of managing forests to achieve one or more clearly specified objectives of management with regard to the production of a continuous flow of desired forest products and services without undue reduction* *of its inherent values and future productivity and without undue undesirable effects on the physical and social environment*" (ITTO 2005). The Ministerial Conference on the Protection of Forests in Europe, held in Helsinki in 1993, established that "*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 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*"

(Ministry of Agriculture and Forestry-Finland 1993). The Convention on Biological Diversity (CBD) defines sustainable use as "*the use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations*".

Vue the lack of a unified definition, McCool and Stankey (2001,2004) stated that sustainability has become a principle "that cannot be proven or measured but which serves to create a sense of community, connection and purpose". Dudley et al. (2005) consider it "no easier to define than love, hope, and charity". They also consider that translating it into practical guidance requires clear agreement concerning what to sustain. Tools such as the Criteria and Indicators processes (C&I) may play a role (Dudley et al. 2005).

### *Criteria and Indicators (C&I) Processes.*

Forest management is a very broad term. It encompasses practices that can range from clear-cutting a whole forest to simply leaving it as it is without cutting any tree. Certainly, some practices and management schemes are more sustainable than others. However, forest managers lack the means for establishing if the management they are giving to a given forest area is sustainable or not apart from the productive perspective. Based on this, several international initiatives have undertaken the development of sets of principles[3], criteria[4], indicators[5] and

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[3] A fundamental truth or law as the basis of reasoning or action. (CIFOR 2000).

[4] A standard that a thing is judged by. A 'second order' principle, one that adds meaning and operationally to a principle without itself being a direct measure of performance. (CIFOR 2000).

verifiers[6] (C&I processes) (Figure 1.1) to be used by managers as tools for directing their practices towards sustainability within a given timeframe and with the ability to monitor trends. They consist of a group of broad core values (Principles and Criteria) that are supported by a number of measures (Indicators and Verifiers) to assess the status or progress towards the realization of these values (Halle 2001). These initiatives include a wide variety of indicators that must be assessed to build an integrated view of sustainability (Stork et al. 1997; Wijewardana 1998; CIFOR 2000).

C&I allow organizing information in a manner that is useful in conceptualizing, evaluating and implementing sustainable forest management (Prabhu et al 1996). They provide a science supported framework upon which management and political decisions can be based (Hall 2001).

C&I processes have become very popular. Since 1992 more than 100 countries in six continents have participated in the development of C&I (Wijewardana et al. 1997; Pokorny and Adams 2003). Holvoet and Muys (2004) were able to compare 164 different standards for SFM under development at that time. Using multivariate statistics and a generic set of principles, criteria and indicators put together considering all of the proposed in all the sets, while discarding the redundant ones; they found that differences between sets are mainly based on the level of application. Those to be used at a National level are more general and are based on others used at regional (sub-national) or at the Forest Management Unit (FMU)[7] level. These last ones are the most detailed and closely related with forest management practices. There are also

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[5] Any variable or component of the forest ecosystem or management system used to infer the status of a particular criterion.

[6] Indicators are evaluated by means of *Verifier*s: They are actual data collected in the field. They provide specific details that would indicate or reflect a desired condition of an indicator, criteria and principle. They may define the limits of a hypothetical zone from which recovery can still safely take place (performance threshold/target). On the other hand, they may also be defined as procedures needed to determine satisfaction of the conditions postulated in the indicator concerned (means of verification). (CIFOR 2000).

[7] A Forest Management Unit (FMU) is a forest area that is being managed under a plan that has been formulated under specific objectives based on existing social, ecological, topographic and productive conditions. Each FMU constitutes an independent management unit.

differences due to the geographic origin of the sets. Biophysical and socio-economic differences influence the scope of the standards. Developing countries standards emphasize social and economic aspects of sustainability whereas developed countries emphasize ecological forest functions. One of the sets best accepted for tropical forests FMU's is being developed since 1994 by the Center for International Forestry Research (CIFOR) (Pokorny and Adams 2003).

### *The CIFOR Process*

The CIFOR was established in 1994 as a member of the CGIAR (Consultative Group in International Agricultural Research) group of research institutions established by the United Nations (UN) in response to the global concern about the social, environmental and economic consequences of loss and degradation of tropical forests. Since then, CIFOR has established itself as one of the principal research institutes involved in C&I development. The CIFOR C&I generic template (CIFOR 2000) constitutes a comprehensive set of C&I assembled by a series of expert teams based on results of research in forest ecology, conceptual frameworks, and field evaluations. This generic template allows addressing how political, ecologic, economic, and productive aspects of forest management affect the sustainability of a management operation (Prabhu et al. 1996; Stork et al. 1997; Mendoza and Prabhu 2000, 2003, 2004, 2005; CIFOR 2000; Pokorny and Adams 2003). Concern was most commonly raised about the practicality of the proposed indicators, or their relevance to forest management. Measurements should be quick, adaptable to the specific conditions of each management operation and relatively inexpensive if C&Is are to be adopted by forest managers and governments (Stork et al. 1997).

CIFOR's set of C&I for ecological sustainability (Appendix 1), considered a "work in progress" as it is in constant improvement, was originated mainly from research made by CIFOR on indicators for conservation of biological diversity in managed forest systems. Focus was on the processes that maintain biodiversity in sustainably-functioning landscapes as it was considered the most cost-effective approach to assessing biodiversity. It has been tested under field conditions in Indonesia, Cameroon, USA (CIFOR 2000), Nicaragua and Costa Rica (Aguilar-Amuchastegui 1999, Mendoza and Prabhu 2000, 2001, 2002, 2003, 2004; McGinley and Finegan 2003; Finegan et al. 2004). The hierarchical structure, moves from a landscape scale down to FMU, community, species, and the genetic levels. The approach is based mostly on the Biodiversity – Sustainability relationship; the second one being a surrogate of the first as biodiversity determines the structural and compositional aspects affecting the dynamics and functioning of a given forest (Noss 1990, 1999; Mankin 1998; Stork et al. 1998; Finegan et al. 2004).

Most of the indicators identified to be used at FMU level are evaluated through verifiers that must be surveyed and measured in the field (CIFOR 2000; Finegan et al. 2004). This makes surveys extremely expensive, time-consuming and logistically challenging (Farthing et al. 2001; Fahrig 2003). As the total area under management and/or the number of FMUs increase, logistical constraints also increase, making it impossible to assessing all of the areas under management. A prioritization system becomes necessary. Yet there are currently no generic approaches to constructing it (Finegan et al. 2004; Aguilar-Amuchastegui 1999, Davis et al. 2001; Fahrig 2003).

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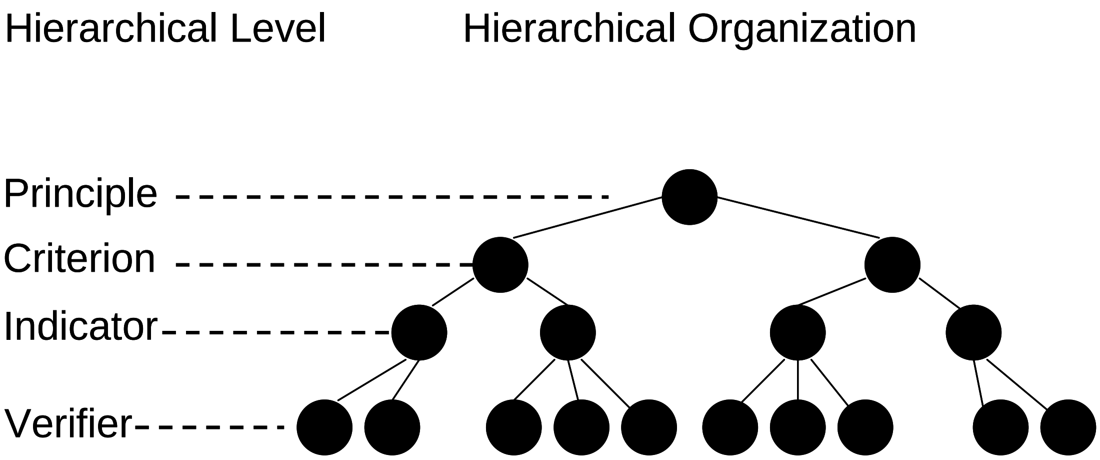

**Figure 1.1.** Hierarchical organization of CIFORs C&I

# Chapter 2: A well known ecological sustainability indicator: dung beetle community structure and composition

Dung beetles are insects from the order Coleoptera, families Scarabaeidae, Geotrupidae and Aphosdiidae (*sensu* Hanski and Cambefort 1991). As their name indicates, they generally feed on dung even though some are copronecrophagous (Davis 1996; Andresen 2002). Their use of feces has also reproductive purposes (Halffter and Edmonds 1982; Hanski and Cambefort 1991). Once dung beetles detect dung, they usually relocate it by burying it under the soil. Some of them just dig tunnels underneath the dung pile; others roll portions of it as part of a courtship display before mating (Hanski and Cambefort 1991). This behavior has important ecological consequences such as the removal of dung from the soil (Gillard 1967; Tyndale-Biscoe 1994), it improves soil fertilization and aeration (Andresen 2002), it increases nutrient cycling rate (Nealis 1977), it helps reduce populations of disease-causing organisms such as flies and hookworms by competing for food resources and destroying eggs and larvae (Hanski and Cambefort 1991; Bergstrom et al. 1976) and they play an important role in secondary seed dispersal reducing predation rates (Estrada and Coates-Estrada 1991; Shepherd and Chapman 1998; Feer 1999; Vulinec 2000; Andresen 1999, 2001, 2002; Wehncke et al. 2004; Wehncke and Dalling. 2005; Gross-Camp and Kaplin 2005; Martins 2006; Vander Wall et al. 2005; Scheffler 2005).

Dung beetles rely on dung production. This links them with forests vertebrate fauna; especially with mammals (Estrada et al. 1999; Castellanos et al. 1999; Pineda et al. 2005; Vernes et al. 2005). Some species are highly sensitive to habitat disturbance showing clear preferences for specific environments such as forest interior, gaps and edges (Nealis 1976; Klein 1989; Halffter and Favila 1993; Hill 1996; Halffter and Arellano 2002; Vernes et al 2005), while others show a wide spectrum of habitat preferences (Scheffler 2005). This makes the organization of their community very sensitive disturbances in microclimate and vegetation structure (Davis et al. 2001; Andresen 2005), soils, and food abundance and distribution patterns (Fincher 1973; Escobar and Ulloa; 2000; Medina et al. 2002; Escobar et al. 2005). Based on this, dung beetle community structure and composition has been proposed and used as indicator of habitat disturbance, particularly in tropical rainforest (Halffter and Favila 1993; Favila and Halffter

1997; Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000; Davis et al. 2001; Pineda et al. 2004; Scheffler 2005; Andresen 2005).

In addition to their ecological characteristics, dung beetle community structure and copmposition fills most of the basic requirements in order to be considered a good indicator: (1) they are taxonomically diverse which means the group has within it a variety of species with different levels of sensitivity to different types of disturbance allowing the use of the whole community as indicator, (2) they have a relatively well known taxonomy, (3) there is abundant information on their natural history, (4) they can be easily sampled and sorted, (5) as mentioned before, some species show high ecological fidelity, (6) they are functionally important within the ecosystems were they are present and, (7) they exhibit a rapid response to environmental changes (McGeoch et al. 2002; Scheffler 2005; Pineda et al 2004). This has resulted in their consideration for use as verifiers in forest management sustainability Criteria and Indicators Processes (Aguilar-Amuchastegui et al. 2000; Finegan et al 2004). However, as mentioned in Chapter 1, in order to be used by forest managers, practical approaches must be developped (Stork et al. 1997).

Dung beetle community structure ,composition and diversity has been found to be a viable assessment tool for forest management ecological sustainability assessment when used in conjunction with standard forest stand management data in tropical lowland forests in Costa Rica (Aguilar-Amuchastegui-FUNDECOR *unpublished data*). Forestry workers have been able to conduct field surveys after basic training in beetle taxonomy and managers (mostly forest engineers) have been able to interpret and incorporate the information provided by the results observed in each FMU to make management decisions to maintain forest productivity while conserving forest biodiversity and functionality. At present, forest workers and managers with which Aguilar-Amuchastegui worked (see Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al 2000) are providing survey services to other forest management operations**.** Both dung beetle community structure, diversity and composition and the assessment methods used, are being proposed by the World Wildlife Fund (WWF) and the Tropical Agronomic Research and Higher Education Center (CATIE) for their use in Forest Stewardship Council (FSC) forest certification surveys in Central America (Finegan *et al.* 2004). Yet, as forest area under management has increased dramatically (from 8000 to 43,000 ha between 1999-2006), so have the logistical constraints for management in general. As basic production-related operations are considered prioritary, the continuity of the ecological sustainability survey program is now under discussion (FUNDECOR, personal communication). *The question of how to assess these indicators, where logistical constraints become critical, remains to be answered from an operational viewpoint. Priorization is necessary.*

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# Chapter 3: Forests structural heterogeneity: a possible approach to prioritization

As mentioned in chapter 1, the CIFOR set of Criteria and Indicators are structured hierarchically in a way that follows the hierarchical structure of biodiversity (see Appendix 1 and Figure 1.3.). The methodology for their application starts the assessment in the same way: those C&I pertaining to a broad scale (landscape level) are evaluated before proceeding to more specific levels, such as Forest Management Unit, Community, Species, and finally the genetic level (CIFOR 2000). Higher levels may then be used as means of prioritizing those areas in which more detailed surveys are necessary. Only those areas that show significant changes in the higher levels may be considered as priorities for the evaluation of more detailed indicators and verifiers.

In the case of ecological sustainability, the main principle identified is the *"maintenance of ecosystem integrity"*. One of the two related criteria is criterion C.2.1.: "*The processes that maintain biodiversity in managed forests (FMUs) are conserved*". Between such processes, the disturbance regime is considered as a key factor as tropical forests are not steady-state ecosystems. Tropical forests exhibit a range of natural disturbance levels. Their historical range of variability (cf. Morgan et al. 1994) embraces a dynamic of disturbances in addition to gap dynamics and creates a mosaic of habitats differing in microclimate, vegetation structure, and faunal composition (Figure 3.1) (Whitmore 1991; Huston 1994; Grieser johns 1997; Charles-Dominique et al. 1998; Hill et al 2001; Rosenzweig 2003). The distribution of these patches generates the vertical and horizontal heterogeneity of a forest.

Depending on its intensity (viz., the number of trees or the basal area or of cubic meters of wood removed per ha), management can change forest structural heterogeneity (Macedo & Anderson 1993; Medley 1993; Vermeulen 1996; Delgado et al. 1997; Bhat et al.; 2000; Chittibabu & Parthasarathy 2000; Bhuyan et al. 2002; Finegan et al. 2004). The more intense the intervention, the more pronounced the changes are (Delgado et al. 1997; Guariguata 1998; Morgan et al. 1994; Ghazoul 2002; Ghazoul and Hellier 2000). Accordingly, CIFOR (2000), established as one of its indicators "*The change in diversity of habitats as a result of human interventions are maintained within critical limits as defined by natural variation and/or regional conservation objectives* "( I.2.1.2 ) with its corresponding verifiers:

V.2.1.2.1. *Vertical structure of the forest is maintained within natural variation*
V.2.1.2.2. *Size class distribution does not show significant change over natural variation.*
V.2.1.2.3. *Frequency distributions of leaf size and shape are maintained within natural variation.*
V.2.1.2.4. *Frequency distribution of phases of the forest regeneration cycle is maintained within critical limits.*
V.2.1.2.5. *Canopy openness in the forest understorey is minimized.*
V.2.1.2.6. *Other structural elements do not show significant change.*
V.2.1.2.7. *The distribution of above ground biomass does not show significant change as compared to undisturbed forest.*

Therefore, if a management scheme maintains the relative abundance and distribution of the successional patches that provide a forest its diversity of habitats, within critical limits established by natural variability, management may be considered sustainable. In contrast, finding a forest that differs in its structural heterogeneity before and after harvesting, after a reasonable recovery time, or in comparison with a natural control area, would constitute *prima facie* evidence of a likely unsustainable management.

*Remote sensing of forest structure*

Several studies have demonstrated how forest structural heterogeneity is related with the diversity of species occurring within a given forest (Charles-Dominique 1998; Hamer et al. 2003, 2005; Vieira et al. 2004; Clarke et al. 2005a,b). This makes forest structural heterogeneity a potential candidate use for identifying priority areas for the evaluation of more specific indicators like those established to be used at guild, species, population and genetic levels (Appendix 1: Indicator I.2.1.3. and below).

However, as it occurs with dung beetle community structure and composition and most of the indicators and verifiers for ecological sustainability, forest heterogeneity verifiers (e.g. structural parameters) are usually surveyed in the field (Finegan et al 2004), which does not really bring any solution to the issue of practicality. A different approach for surveying is required.

The heterogeneity of forest structure can be measured by means of remote sensing (Wulder 1998a,b; Lim et al.2003; Lambin 1998; Foody and Cutler 2003; Read et al. 2003; Lu et al. 2004; Kalácaska et al. 2004; Wulder et al. 2004; Asner et al. 2005a; Couteron et al. 2005; Ingram et al. 2005), allowing the estimation and monitoring of global carbon stocks (Couteron et al. 2005) and, the retrieval of spatial information on potential determinants of plant and animal species distribution, (Couteron et al. 2003; Clarke et al. 2005a), gap-phase regeneration stages and, soil types and conditions (Ashton and Hall 1992; Tuomisto et al. 2003). The reflectance pattern of a forest is determined by the structure of the canopy (Danson 1995), understory leaf litter (Franklin et al. 2002), biomass, age, density, mean tree height and basal area (Lee & Nakane, 1996; Lu et al 2004).

Remote sensors such as the Landsat TM and ETM+, SPOT, Laser Vegetation Imaging Sensor, LiDAR, and Synthetic Aperture Radar: (JERS-1, JERS-2, JPL/AIRSAR, ERS-1, ERS-2, ENVISAT, RADARSAT), have proven to be useful for estimation of forest stand variables such as biomass, basal area, Leaf Area Index (LAI), average stand diameter and average stand height (Wulder 1998a; Lim et al.2003; Lu et al.2004; Israelsson and Askne 1995; Pulliainen et al. 2003; Lim et al. 2003; Santos et al. 2003; Feeley et al. 2005; De Wasseige and Defourney 2002, 2004; Asner et al. 2002, 2004; Lu et al. 2004; Feeley et al. 2005; De Wasseige and Defourney 2002, 2004). Such findings suggest that remote sensing could be used in their use for evaluating those CIFOR Indicators that are assessed by means of verifiers related to forest structure and stand variables; more specifically indicators I.2.I.I. *Landscape pattern is maintained* and I.2.I.2. *The change in diversity of habitat as a result of human interventions are maintained within critical limits as defined by natural variation and/or regional conservation objectives.* (see Appendix 1).

The wide variety of sensor spatial resolutions allows their use for verifiers working at different spatial scales. Csillag (2000) proposed a simple classification of sensors to be used according to the spatial scale of the area or phenomena studied. He proposed to use the relationships between observational and ecological characteristics scales. In this case, the spatial scale at which each specific verifier is assessed and the spatial resolution of the sensor (Figure 3.2). Sensors such as the Landsat TM, ETM+ and SPOT and more recently EO-1 ALI have proven to be the most versatile for forest stand parameters estimation as their spatial resolutions (between 20-30m) are consistent with the average quadrant size used when surveying in the field (Lu et al 2004, Asner et al. 2002, 2004, Finegan et al. 2004).

### *Vegetation structure and vegetation indices*

There are several methods for utilizing and analyzing spectral information to assess vegetation structural variables (Ingram et al. 2005). One of the most common approaches is combining the spectral information from multiple bands (blue, green, red, near infra-red) (Figure 3.3) into a composite value known as a spectral vegetation index (VI). VIs have been shown to be related with vegetation structural and biophysical characteristics such as Chlorophyll content, Net Primary Productivity (NPP), Leaf Area Index (ALI), and the amount of photosynthetically active radiation that is incoming into a canopy that is being absorbed (FPAR) which in turn are related with forest structure and successional, phenological and physiological stages (Rouse et al. 1974; Danson 1995; Gitelson 2004; Gitelson et al 2003, 2005, 2006; Cohen and Goward 2004; Kalácaska et al. 2004; Viña and Gitelson 2005). Most vegetation indices use the contrast between the reflectance levels in the red and the near infra-red portions of the spectrum: a strong absorption in the red and a strong reflectance in the near infra-red. As biomass and related vegetation variables increase so does the contrast (Kalácaska et al. 2004).

The Normalized Difference Vegetation Index (NDVI) (Rouse et al. 1974) [1.1] is the most commonly used vegetation index (Ingram et al 2005) as it shows good relationships with vegetation structure and biophysical characteristics (Baret & Guyot 1991; Henebry 1993; Goodin & Henebry 1997; Guillevic & Gastellu-Etchegorry 1999; Gitelson et al. 2003; Gitelson 2004; Asner et al. 2004; Kalácaska et al. 2004, 2005a,b; Shabanov et al. 2005). It is formulated as:

$$NDVI = \frac{\rho_{NIR} - \rho_{RED}}{\rho_{NIR} + \rho_{RED}} \qquad [1.1]$$

where $\rho_{NIR}$ and $\rho_{RED}$ correspond to the reflectances measured in the near infrared and the red portions of the spectrum (Figure 3.3).

However, NDVI loses sensitivity when a certain threshold of a given vegetation biophysical variable is reached. Its value can remain unchanged as increments in biomass or in related biophysical parameters occur (Figure 3.4) (Baret & Guyot 1991; Gitelson 2004; Vina et al. 2004; Kalácaska et al. 2004). This is common case in tropical forests where NDVI and other indices such as the NLI, MSR, SAVI2 SR or IRI have not shown a good relation with mature forest structure (Lu et al. 2004; Kalácaska et al. 2004; Kalacska et al. 2005a,b). An alternative index: the Wide Dynamic Range Vegetation Index (WDRVI) [1.2] was developed by Gitelson (2004) for use with denser vegetation, (with LAI values of 5 and up) such as tropical forests (Cournac et al. 2002; Gitelson 2004; Vina et al. 2004; Vina and Gitelson 2005; Gitelson et al. 2005). The Wide Dynamic Range Vegetation Index (WDRVI) is defined as:

$$WDRVI = \frac{\alpha\rho_{NIR} - \rho_{RED}}{\alpha\rho_{NIR} + \rho_{RED}} \qquad [1.2]$$

where $\rho_{NIR}$, is the reflectance measured in the near infrared zone of the electromagnetic spectrum, $\rho_{RED}$ is the reflectance level measured in the red zone and, $\alpha$ is a weighting coefficient for $\rho_{\mathrm{NIR}}$ .

The value of $\alpha$ can decrease $\rho_{\mathrm{RED}}$ and thus improve the sensitivity of WDRVI to changes in vegetation biophysical parameters. This enhances the dynamic range of the NDVI, improving the sensitivity of WDRVI to changes in vegetation biophysical variables when vegetation is dense (Gitelson 2004) (Figures 3.4 and 3.5). Note that if $\alpha$=1 then WDRVI=NDVI. In order to obtain the optimal enhancement of WDRVI's dynamic range, the appropriate alpha value needs to be established (see Henebry et al. 2004).

*Remote sensing of vegetation heterogeneity*

The rationale behind the use of VIs for remote surveying of forest structural heterogeneity lies in the extraction of spatially explicit information. Both, spatial location and index values provide a tridimensional perspective of the vegetation being surveyed (Figure 3.6). The study of the spatial pattern of vegetation indices will provide information about forest structure spatial patterns (heterogeneity) (Henebry 1993; Goodin and Henebry 1997; Goodin et al. 2004).

Wagner and Fortin (2005) defined spatial heterogeneity as “the spatially structured variability of a property of interest which may be categorical or quantitative”. It occurs when a variable (e.g. forest structure) observed at different locations shows variation with a spatial dependence. Variation comes with distance conducing to the existence of gradients making the landscape to be heterogeneous (Atkinson 1997).

Among the methods for quantifying spatial heterogeneity, geostatistical tools such as variograms and autocorrelograms are used most (Isaaks and Srivastara 1989; Légendre and Fortín 1989; Riera et al. 1998; Duteuil and Légendre 1993; He et al. 1996; Pastor et al. 1998; Goodin et al. 2004; Colombo et al. 2004; Zawadski et al. 2005). They provide of measures of both dimensions of heterogeneity: variation and scale (Henebry 1993; Goodin and Henebry 1997; Goodin et al. 2004).

Variogram analysis is ideal for assessing spatial heterogeneity as it provides a graphic representation of the variation of a given characteristic (in this case, a vegetation index) as a function of distance lags (StOnge and Cavayas 1995; He et al 1996; Atkinson and Tate 2000; Goodin et al. 2004; Colombo et al. 2004; Zawadzki et al. 2005). There are three basic parameters: *The range* indicates the distance where the semivariance ceases to increase and sampling units become non spatially correlated, providing a representative sample of the total variance observed within the specific forest area. *The sill* corresponds to the semi-variance registered by the variogram when it reaches the range distance (scale). Finally, the *nugget variance* corresponds to the variance registered at distance zero, and represents the local variation occurring at scales finer than the variogram sampling interval ($h$), due to fine scale variability, and/or measurement error. This is called “the nugget effect” (He et al.1996, see Légendre and Fortin 1989; Duteuil and Légendre 1993; Légendre and Légendre 1998). Both, the sill and the nugget contribute to the total variation $\gamma(h)$ measured within a given forest area. Overall, total variation and range, the two dimensions of the semivariogram, as obtained from VI data would tell us how much structural heterogeneity there is in a given forest area and, across how much space it occurs (Figure 3. 7) (Légendre & Fortin 1989; Duteuil & Légendre 1993; He et al. 1996; Légendre & Légendre 1998). A comparison of the dynamic changes of these variables in both natural and managed forest areas would provide a basis for assessing forest management impacts in forests structure heterogeneity.

In order to establish the magnitudes of range, sill and nugget, the semivariance has to be modeled as a function of lag distance (Isaaks and Srivastara 1989). There are three basic model types commonly used: Spherical, Exponential, and Gaussian. Range, sill and nugget variance are the parameters for which coefficients need to be estimated in order to establish the fit (Woodcock and Harvard 1992; Woodcock et al 1988a,b, 2001)(Table 3.1). The most commonly used is the spherical model as it provides of a better visual fit and a real sill value (Deutsch and Journel 1998). Its interpretation is also more straightforward when working with satellite images data as Woodcock et al (1988a,b) demonstrated. The sill represents a characteristic patch size in the data (i.e. spatial dependence or, in our case, the minimum representative sample of total heterogeneity observed in VI values/forest structure heterogeneity within a given forest area); whereas sill+nugget variance values are the actual measure of spatial heterogeneity (the amount of structural heterogeneity observed within a given forest area) (Goodin et al. 2004). Parameter coefficient estimation can be made using conventional statistic methods such as Indicative Goodness-of-Fit (IGF) (Panatier 1996) seeking to minimize estimation error.

Semivariogram parameter coefficient estimates have been shown to be related to forest types and with biophysical aspects such as vegetation density, stratification, percentage of understory cover, tree crown size, leaf area index (LAI), tree height, and forest structural damage (Cohen et al 1990; St-Onge and Cavayas 1995; Bruniquel-Pinel and Gastellu-Etchegorry 1998; Wulder 1998a,b; Levesque and King 1999; Treitz and Howarth 2000a,b; Wallace et al. 2000; Wallace and Marsh 2005). For example, Cohen et al (1990), established that semivariogram sill values were related to forest vegetation vertical stratification and percent of canopy cover while range values were related to mean tree canopy sizes. Woodcock et al (1988a) established relations between the sill and area covered by objects (say vegetation cover), and the range and the size of the objects (say tree size). Such relationships have enabled their use for improving the accuracy of LAI estimates (Franklin et al. 1996), vegetation cover classification (Miranda et al. 1992, 1996; 1998; Franklin and McDermid 1993; Carr 1996; Franklin et al. 1996; Chica-Olmo and Abarca-Hernandez 2000; Jakomulksa and Clarke 2001) and forest inventorying (Lefsky et al 1999; Hudak et al. 2002).

In order to make a good use of semivariogram analysis its directionality needs to be considered. A semivariogram can be performed in specific directions, which allows its use to find spatial trends and directional gradients within the data (anisotropy) (Deutsch and Journel

1998). This means that when using it to estimate whole forest area parameters, the analyses have to be performed considering all directions (omnidirectional semivariogram) so no biases or loss of information on forest structural heterogeneity occurs.

### *Semivariance of vegetation indices*

Semivariance analysis of vegetation indices and spectral bands combinations has been proposed for its use as ancillary data in forest cover classification procedures (Miranda et al. 1992, 1996; Franklin and McDermid 1993; Franklin et al. 1996), LAI estimation (Wulder et al. 1998) and for assessing forest landscape structure (Ricotta et al. 2003). However, such analysis needs to take into account those parameters of remote sensing that will affect the quality and comparability of the results (Zawadzki et al 2005). Aspects such as the spatial resolution of the sensor, elevation and illumination incident angle (e.g. sun elevation and azimuth angle) and sensor view angle affect semivariance model parameter estimations (Civco 1989; Guyot et al. 1989; Goodin et al., 2004; Zawadzki et al 2005). Goodin et al. (2004), found that when the sensors are in nadir looking position (looking straight down), solar illumination angle can affect sill+nugget (measured total semivariance) but does not affect range coefficient estimates when measuring vegetation structure heterogeneity. In the case of satellite remote sensing data, images must be acquired acquired in similar dates in the year, times in the day, and latitudes in order to be able to control for biases caused by differences in sun elevation and illumination angle.

The spatial resolution of the sensor is related to the semivariogram model parameters. Sensors like the Landsat TM/ETM+ and SPOT (30 and 20m respectively), have been identified as optimal for forest areas, while sensors with higher spatial resolutions are required for individual tree variables (Strahler et al 1986; Fazakas and Nilsson 1996; Wulder 1998a,b; Tomppo and Czaplewski. 2003; Zawadzki et al. 2005)

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**Figure 3.1.** Examples of natural forest structure dynamic stages with canopy view from the understory: **a**) and **b**) natural tree-fall gap (1 day old); **c**) and **d**) mature forest. (Photos: Thomas Perea Bruckner, Finca Agricola Ladrilllera S.A. No 2, Costa Rica, June 2005).

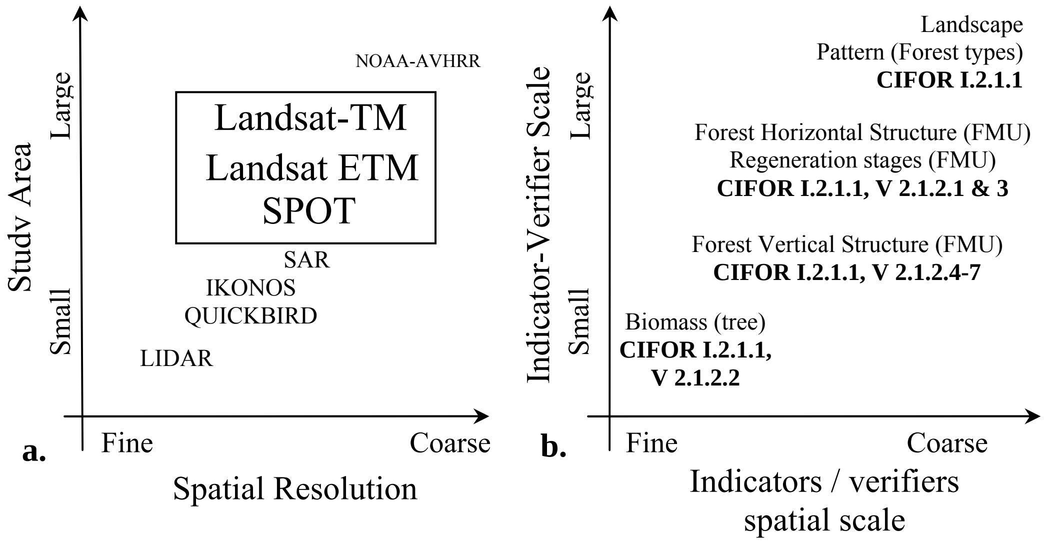

**Figure 3.2.** Relation between sensor spatial resolutions (a) and, CIFOR indicator/verifier spatial scales (b). (modified from Csilllag et al. 2000).

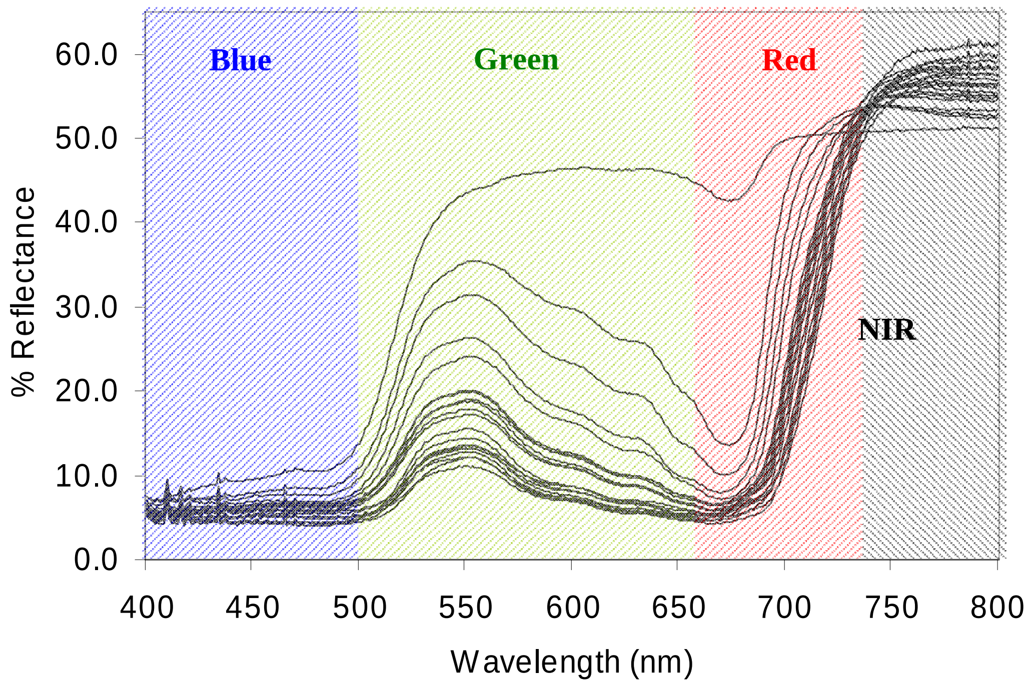

**Figure 3.3.** Reflectance spectra obtained from soybean leaf stacks with different chlorophyll content. Measurements were made using an Oceanoptics USB 2000 sensor. Blue, Green Red and NIR refer to the blue, green, red and near infra-red portion of the electromagnetic spectrum (measurements made by myself in CALMIT's spectroscopy lab). As chlorophyll content increases reflectance of the visible portion of the spectrum (blue, green and red) decreases and the near infrared (NIR) increases.

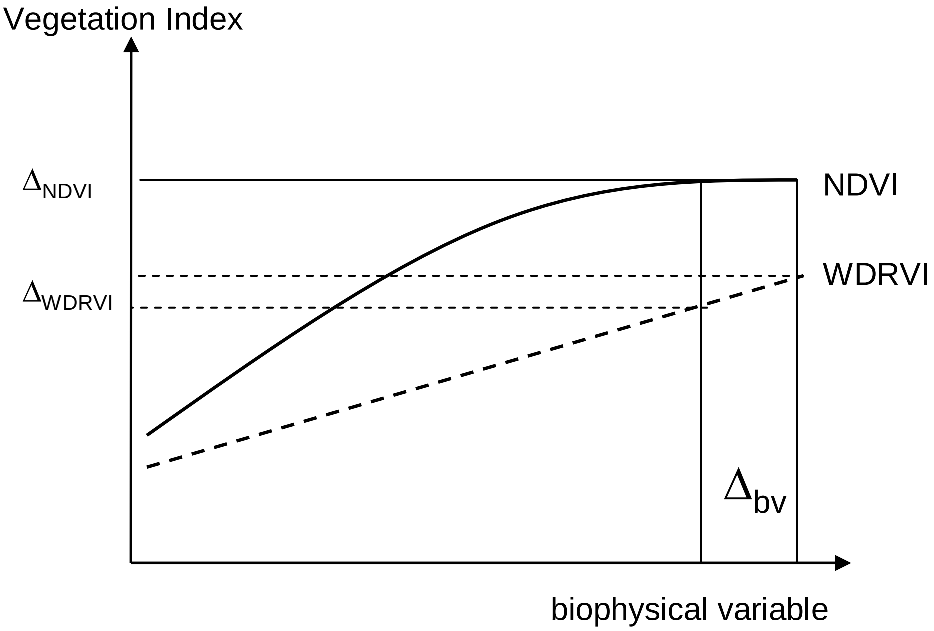

**Figure 3.4.** Relation of the Normalized Difference Vegetation Index (NDVI) and the Wide Dynamic Range Vegetation Index (WDRVI) with a given vegetation biophysical variable. After reaching a certain threshold, NDVI loses sensitivity and remains unchanged while WDRVI continues to change.

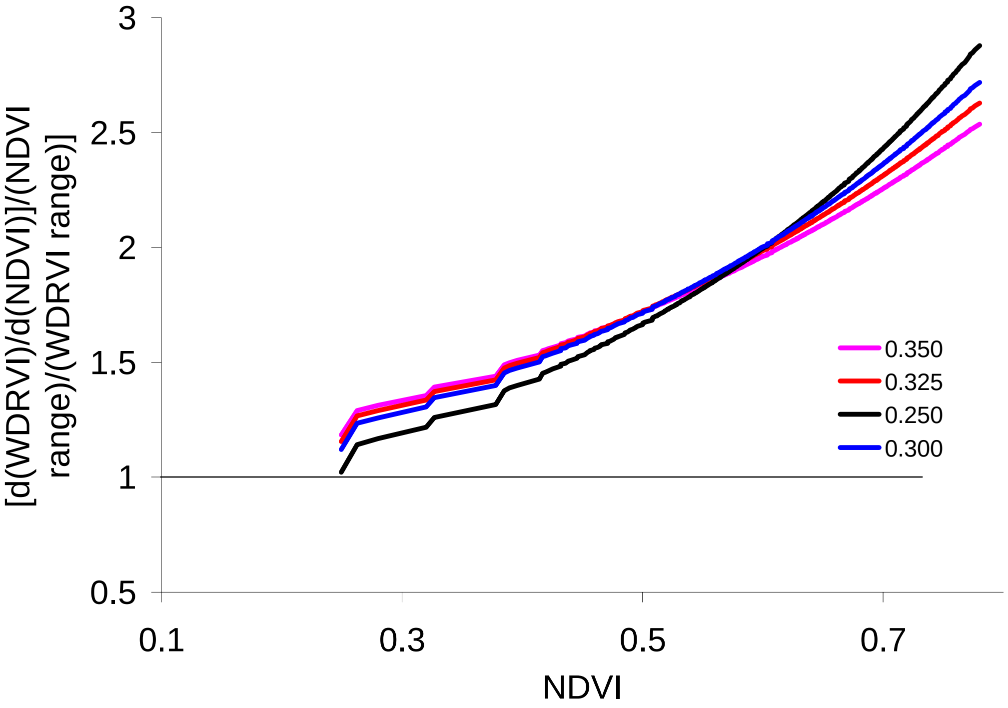

**Figure 3.5.** Sensitivity comparison between NDVI and WDRVI α= 0.25, 0.3, 0.325 and 0.35, as obtained with reflectances measured from Costa Rican tropical forest canopies. When [d(WDRVI)/d(NDVI)]/(NDVI range)/(WDRVI range)] is below 1, NDVI is more sensitive than WDRVI and vice versa. See how WDRVI is more sensitive in all the cases (2001Gitelson et al. 2005; see Gitelson 2004) (Reflectance data from six natural and fifteen managed forest areas obtained from a Landsat 5 TM image [WRS-2 Path 15, Row 53] from 14 January 2001).

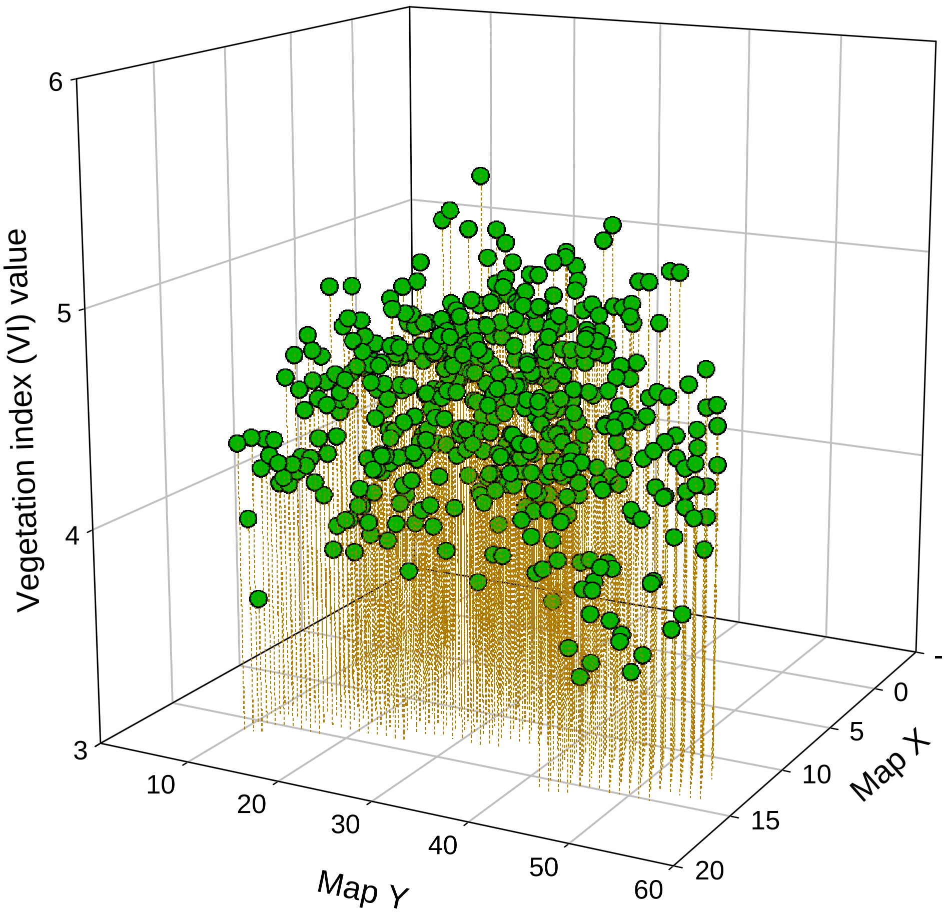

**Figure 3.6.** Three-dimensional perspective of a natural tropical forest area provided by a Vegetation Index (VI). Each dot corresponds to the location and VI value as obtained from reflectance data obtained from a 30mx30m pixel of a Landsat TM image.

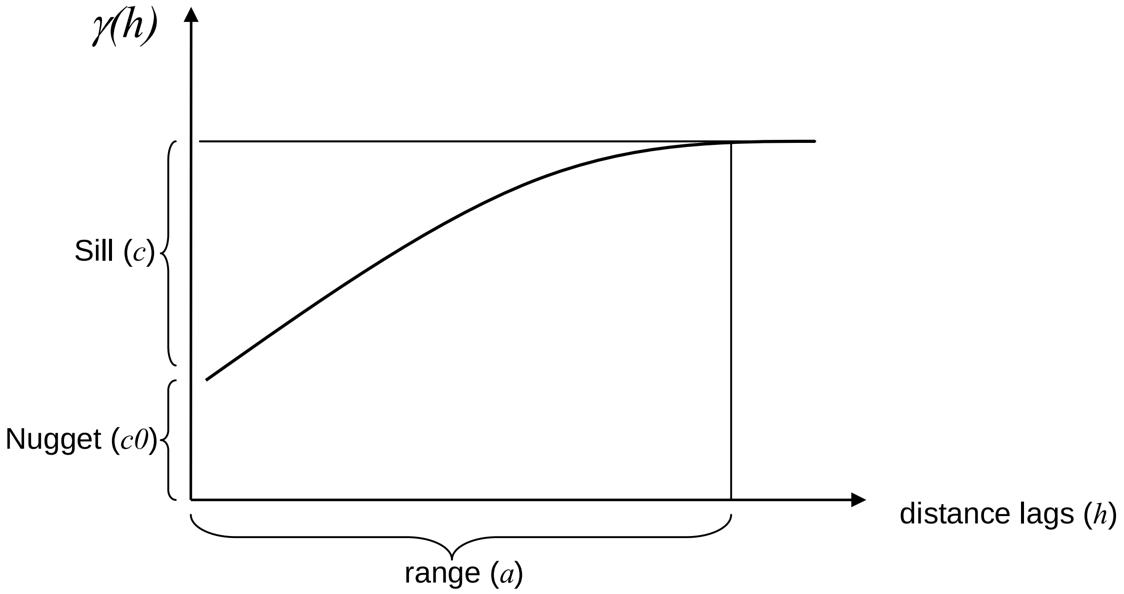

**Figure 3.7.** Semivariogram model with its basic parameters: range, sill and nugget variance. The "x" axis represents the lag distance *(h)* at which occurs the semivariance $\gamma(h)$ ("y" axis). As the lag distance increases, semivariance increases until it reaches the range distance (*a*), and stabilizes itself reaching the sill (*c*). Variance occurring when distance is zero corresponds to the nugget ($c_0$). Both sill and nugget contribute to total variation ($\gamma[h]$).

**Table 3.1.** Models commonly used for semivariogram representation: $h_i$ corresponds to the distance in the specific $i^{th}$ lag, *a* corresponds to the range distance and, *c* is the sill (the semivariance when the range distance has been reached) and, ($c_0$) is the nugget or the variance for a lag distance, *h*=0 (Woodcock et al. 1988a,b).

| Model | Formula |
|---|---|
| Spherical | $\gamma(h_i) = \begin{cases} c_0 + \left[ c.1.5\left(\frac{h}{a}\right) - 0.5\left(\frac{h}{a}\right)^3 \right] & if\ h \le a \\ c_0 + c & otherwise \end{cases}$ |
| Exponential | $\gamma(h_i) = \begin{cases} c_0 + c\left[ 1 - \exp\left(\frac{-3h}{a}\right) \right] & if\ h \le a \\ c_0 + c & otherwise \end{cases}$ |
| Gaussian | $\gamma(h_i) = \begin{cases} c_0 + c\left[ 1 - \exp\left(\frac{-3h^2}{a^2}\right) \right] & if\ h \le a \\ c_0 + c & otherwise \end{cases}$ |

# Chapter 4: Modeling dung beetle diversity as a function of forest structural heterogeneity: model selection

We have established that (1) there is evidence of relationships between dung beetle community structure and composition and forest structural heterogeneity. (2) forest management, depending on its intensity, can affect forest structure and dung beetle community, and (3) forest structural heterogeneity can be measured and monitored by means of semivariogram analysis of remote sensing data as obtained from satellite images. The challenge is now to find evidence of such interrelations and to identify ways to use them to prioritize forest management units for field surveys of ecological sustainability indicators. We seek to establish relationships between an environmental variable such as forest structural heterogeneity, a management variable such as logging intensity, and one aspect of ecological diversity: dung beetle community structure and composition (Figure 1 in the introduction).

A first approach consists in using matrix correlation analysis (Légendre and Légendre 1998) to explore possible relationships between the semivariance of Wide Dynamic Range Vegetation Index values and dung beetle species diversity as measured form various forest management units. In order to be able to establish the utility of forest structural heterogeneity for prioritizing forest areas for dung beetle field surveys, its explanatory power for dung beetle community structure and composition modeling (and that of other environmental variables) needs to be established. The general idea consists in modeling dung beetle ecological diversity as a function of environmental surrogate variables (environmental diversity or ED).

Modeling species richness is the most common manifestation of this approach: regional scaled regression models are built to relate field-surveyed species richness to remotely mapped environmental variables (e.g. Heikkinen and Neuvonen 1997; Leathwick et al. 1998; Lwanga et al. 1998; Wohlgemuth 1998; Ferrier et al. 2002). However, this approach is not useful when the objective is to maximize the total number of species, and therefore diversity, represented in a set of conservation areas, or for identifying priority areas for field surveys of ecological diversity. What need to be modeled are the compositional dissimilarities[8] existing between areas in order to

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[8] Difference in composition of species between geographically separated areas of the same type of habitat or environment (in our case, between forest areas) (Ferrier et al. 2002).

be able to identify those that will better complement an already existing system of protected areas, effectively increasing the total number of species under conservation (Pressey et al. 1993; Ferrier et al. 2002) or, in our case, to identify those FMU that show more dissimilarities with respect to natural reference areas and hence are prioritary for field surveys of ecological sustainability verifiers.

Ferrier et al. (1999) proposed the use of matrix regression, an extension of the matrix correlation approach (Burgman 1987; Legendre and Legendre 1998), to model crossed compositional dissimilarities (e.g. between dung beetle communities) existing between different areas as a function of Euclidean distance matrices of environmental surrogate variables (Poulin and Morand 1999; Ferrier et al. 1999; Ferrier 2002). However, matrix regression assumes that the relationship between the dependent and the independent variable is linear (in this case, compositional dissimilarities as explained by a dissimilarity index, which usually varies between 0 and 1, as is the case of the Bray-Curtis Dissimilarity Index), which has been proven otherwise (Faith 1997; Légendre and Légendre 1998). On the other hand, matrix regression assumes constant rates of compositional change and species turnover across a given range of an environmental variable. Violations to this assumption are common in real data sets generally because environmental variables are many times measured in arbitrary scales (Whittaker 1977, Oksanen and Tonteri 1995; Simmons and Cowling 1996; Ferrier 2002). Accordingly, Ferrier (2002) proposed the use of log-transformations of compositional dissimilarities and generalized linear modeling (McCullagh and Nelder 1989) in what he calls Generalized Dissimilarity Modeling (Ferrier 2002; Ferrier et al. 2002) (Figure 4.1).

This approach has been proposed and used for identifying and establishing new protected areas in Australia (Ferrier 2002, Ferrier et al. 2002; Faith et al. 2003, 2004). The main reason has been that while data about spatial distribution of biodiversity are usually scarce and biased, as field surveys are so demanding and expensive (Fahrig 2003), GIS environmental data such as elevation, slope, aspect, roads, land-use/land-cover, and hydrology are readily available and may be used as a descriptor of the variability that ecological diversity shows between sites (Ferrier 2002, Ferrier et al. 2002 Faith et al. 2003, 2004; Hortal and Lobo 2005). On the other hand this approach allows taking advantage of the ever improving spatial coverage and resolution of remotely derived data, including abiotic environmental layers, land-cover/land-use data, and as in our case, forest structure data extracted from satellite imagery and forest management data such as logging intensity, number of logging events and years since last logging event (Ferrier 2002, Ferrier et al 2002; Faith et al 2003).

Several authors have expressed doubts about the real explanatory power of ED for estimating current biodiversity. Araujo et al. (2001, 2003) found no differences between the results obtained in biodiversity sampling using ED for site selection than those obtained using a randomized method. Such results may be related with the fact that present ED does no account for the effect that historical events can have over the spatial pattern of biodiversity (Holling 1992; Hortal et al. 2005). Hortal et al. (2005) expand on the difficulty of including such contingency variables as means of providing a dynamic baseline to an ED based approach.

We explored the use of the dynamic change of forest structure heterogeneity that occurred before and after logging, in both natural and managed areas as one of the contingency variables. This would allow accounting for the role of different disturbance histories (natural and those caused by management activities) in explaining the current biodiversity present in each of the forest areas (Holling 1992: Araújo et al. 2001; Araújo et al. 2003).

### *Model selection*

One of the key aspects when modeling is the issue of model selection. Which variables to use as explanatory variables, how many, and which model type are only some of the questions that need to be answered. The main question usually tackled is related to the fit of the model or the bias that estimated values show when compared with actual measured values. Conventional model selection approaches such as backward, forward, or stepwise selection procedures are generally based on hypothesis tests, where at a certain p-value, a variable is included or excluded (Zar 1984, Hosmer and Lemeshow 1989, Afifi and Clark 1996,). Root mean square error is used as a measure of model estimate bias over which models are selected (Hilborn and Mangel 1997). However, this method does not account for the precision of a given model which is provided by the estimation of its parameters. As the number of explanatory variables used increases, the number of parameters to estimate also increases and the model becomes less precise, even though bias might decrease (Mazerolle 2004; Hilborn and Mangel 1997; Burnham and Anderson 2002). A balance between model bias and precision must be found. This refers to the principle of parsimony, which suggests that the simplest explanation is probably the most likely (Burnham and Anderson 2002).The ideal is a model that has a low estimation bias and uses as few explanatory variables (and parameters) as needed (Hilborn and Mangel 1997; Burnham and Anderson 2001).

The Akaike Information Criteria (AIC) [4.1] and associated measures constitute the most straightforward method of model selection addressing the issues of model bias and precision (Hilborn and Mangel 1997; Burnham and Anderson 2001). The AIC is not a hypothesis test. It does not have an α value and does not use notions of significance. Instead, the AIC focuses on the strength of evidence and gives a measure of uncertainty for each model (Mazerolle 2004).

$$AIC_c = -2L(S \mid M_i) + 2K\left(\frac{n}{n-K-1}\right) \qquad [4.1]$$

where $-L(S|Mi)$ is the maximum likelihood (or minimum negative log-likelihood [4.1.1]) of model $i$ obtained with the best fitting parameters, n, is the number of samples, K, is the number of parameters in the model and, $\left(\frac{n}{n-K-1}\right)$, is a sample size bias corrector that may be used to obtain a corrected AIC (see that if $n$ is very large with respect to $K$ the correction factor gets close to 1, so in the end $AIC_c$ for a large sample size corresponds to AIC) (Hilborn and Mangel 1997).

$$L(S|Mi) = n\left[\log(\sigma) + \frac{1}{2}\log(2\pi)\right] + \sum_{i=1}^{n}\frac{(Y_i - m)^2}{2\sigma^2} \qquad [4.1.1]$$

where $L(S|Mi)$ is the negative log-likelihood of model $i$, $n$, is the number of samples, $\sigma$ is the standard error of the measured data, and $(Y_i - \mathrm{m})^2$ are the squared error of the estimates obtained with model $i$. (Hilborn and Mangel 1997).

The AIC value in itself is meaningless. It obtains its meaning when it is compared to the AIC values obtained by a series of other models specified a priori, using different sets of explanatory variables or formulae. The model with the lowest AIC would be the “best” model among all the models specified for the data at hand (Hilborn and Mangel 1997; Burnham and Anderson 2002; Mazerolle 2004). However, if only poor models are considered, the AIC will only help identifying the least poor of all. This highlights the importance of spending time determining the set of variables, and formulae for the candidate models. They must be selected based on previous investigations, as well as on judgment and knowledge of the system under study (Mazerolle 2004).

Two measures associated with the AIC can be used to compare models: the delta AIC ($\Delta_i$) and the Akaike weights ($w_i$). The simplest, the delta AIC [4.2], is a measure of each model relative to the best model. As a rule of thumb, a $\Delta_i < 2$ suggests substantial evidence for the model, values between 3 and 7 indicate that the model has considerably less support, whereas a $\Delta_i > 10$ indicates that the model is very unlikely (Burnham and Anderson 2002).

$$\Delta_i = AIC_i - AIC_{\min} \qquad [4.2]$$

where $AIC_i$ is the AIC value for model $i$ , and $AIC_{\min}$ is the AIC value of the “best “ model.

The Akaike weights ( $w_i$ ) [4.3] provide another measure of the strength of evidence for each model, and represent the ratio of delta AIC ($\Delta_i$) values for each model relative to the whole set of *R* candidate models. This allows rescaling the $\Delta_i$*'s* so the sum of the $w_i$ equals 1. This makes the interpretation of Akaike weights ($w_i$) more straightforward as the weight actually indicates the probability that the model is the best among the whole set of candidate models (Hilborn and Mangel 1997; Burnham and Anderson 2002; Mazerolle 2004).

$$w_i = \frac{\exp\left(-\frac{1}{2}\Delta_i\right)}{\sum_{r=1}^{R}\exp\left(-\frac{1}{2}\Delta_r\right)} \qquad [4.3]$$

All of the model evaluation techniques presented here, were used in the present research in order to establish which environmental variables best explained the dung beetle community compositional dissimilarities found between the forests (natural and managed) that were surveyed in the field. More specifically, We sought to establish the explanatory power of environmental variables derived from the assessment of forest structural heterogeneity by means of semivariogram analysis of vegetation indices obtained from satellite remote sensing data, and a key forest management variable: logging intensity. Additional geospatial data such as elevation, slope, aspect, geographic distance between sites, and geographic distance to natural areas were also considered.

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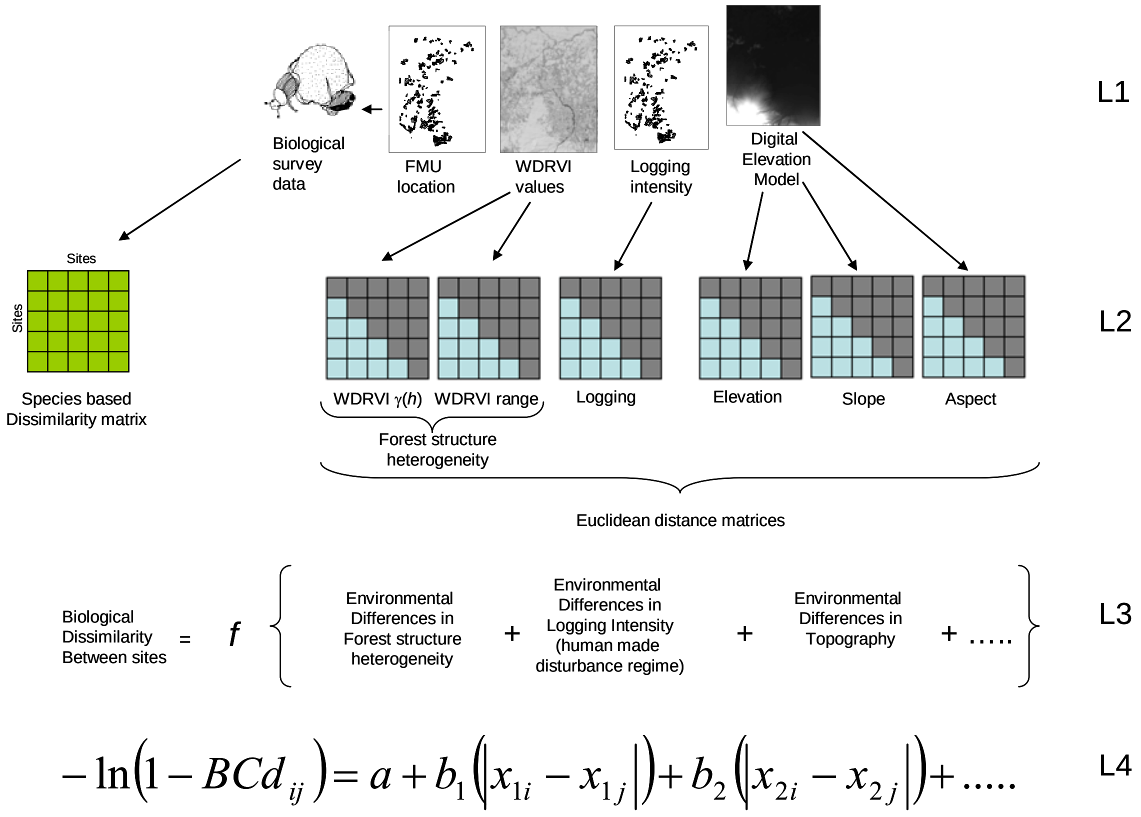

$$-\ln\left(1-BCd_{ij}\right)=a+b_1\left(\left|x_{1i}-x_{1j}\right|\right)+b_2\left(\left|x_{2i}-x_{2j}\right|\right)+\ldots..$$

**Figure 4.1.** Modeling of ecological dissimilarities by mean of environmental surrogate variables (environmental diversity). Level L1 represents the available layers of data, including biodiversity data for each one of the sampling locations. Level L2 represents the ecological dissimilarities matrix obtained based on each community species composition and the Euclidean distance matrices as obtained for each environmental variable between sampling sites. Level L3 represents the general structure of the model which is shown in more detail in level L4 after logarithmic transformation of ecological dissimilarities (adapted from Ferrier 2002).

# Section II.

## Chapter 5: Study area

This study was made in Costa Rica, Central America. Costa Rica, despite its small size (51,100 km$^2$), has high levels of biological diversity with some 12,000 species of plants, 1,239 species of butterflies, 838 species of birds, 440 species of reptiles and amphibians, and 232 species of mammals (INBIO 2006). In 1900, 99% of the country was forested. Today 46.8% of the country remains under some type of forest cover (Table 5.1). Historically, clearing for agriculture (mostly coffee and bananas) and cattle pastures has been the largest contributor to Costa Rica's rainforest destruction. During the 1970s and early 1980s, vast stretches of rainforest were burned and converted into cattle lands. Today, while deforestation rates of natural forest have dropped considerably, Costa Rica's remaining forests still face threats from illegal timber harvesting in protected areas and conversion for agriculture and cattle pasture in unprotected zones (Gonzalez and Lobo 1999; FAO/FRA 2000; McKenzie 2004).

The study area is located in the Canton of Sarapiqui, in the Atlantic slope of the Cordillera Volcanica Central between 20 and 1500 meters above the seal level, near Braulio Carrillo National Park (Figure 5.1). The region is characterized by a fragmented landscape as a consequence of forest clearing for the establishment of commercial crops such as banana and palmito plantations and extensive cattle production (Figures 5.2 and 5.3). Rainfall occurs all year-long in a bimodal pattern showing reductions between February-April (around 200 mm) and September and increases between May-July and November-December (Figure 5.4). (La Selva 2006; Sanford et al. 1994). It is in this area that Fundacion para el Desarrollo de la Cordillera Volcánica Central (FUNDECOR) a non-governmental organization (NGO) founded in 1991 has its main activities (FUNDECOR 2006).

Since its inception, FUNDECOR has been working on the protection of the natural resources of the Costa Rican Central Mountains and its surrounding areas by means of market based sustainable strategies. The conceptual basis of this mission is the principle that conservation and development should coexist in harmony. With this mission in mind FUNDECOR designed the strategy of creating profitable green market alternatives for forest owners. Among the activities implemented by FUNDECOR with forest owners, sustainable forest management is a key element. FUNDECOR provides forest owners with planning, transfer of technology, and technical support services to guarantee the sustainable harvesting of wood (FUNDECOR 2006). At present, FUNDECOR has established more than 450 contracts with small forest dwellers to provide technical assistance to manage their forest under the strictest standards of environmental sustainability. The total extent of the land under FUNDECOR's supervision is 40,000 hectares, providing benefits to 2,600 direct beneficiaries and almost 40,000 indirect beneficiaries. FUNDECOR has also helped the Costa Rican government design a system to manage the payment of environmental services provided by privately owned forests.

### *Forest Management Program*

FUNDECOR forest management program uses forest impact minimization techniques that include extensive use of GIS for planning harvest and monitoring. Each forest management unit is characterized in terms of topography, hydrology, trees to be cut, and preserved locations (Figure 5.5). Based on this information, hauling roads and erosion and water protection areas are established in order to minimize logging impact (Figure 5.6). As soon as the forest is harvested, drainage systems are built in order to reduce soil erosion and enhance recovery (Figure 5.7). Figure 5.6 shows an example of a standard forest management unit map as conceived based on its specific management plan. It shows the productive area, slope (+75%) and hydrologic restriction areas, the streams, inventory transects, hauling trails, permanent plots, the location of trees to be cut with specific fall direction cones. Trees supposed to remain as reproductive sources, non-commercial trees, and trees under minimum cut diameter are also shown. With this map, logging crews can navigate the unit and perform all logging activities within the management plan guidelines established based on the FMU specific conditions.

After logging has taken place forest recovery is assessed by means of sustainability criteria and indicators evaluations (FUNDECOR 2006; Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000), and based on forest inventory results, the cut cycle length in years is established (Figure 5.8). At present, a total of 10,024 hectares of natural forests have been covered with this program.

Imagery with low cloud cover is available for this area from a variety of sensors (Figure 5.9). Additionally, all FUNDECOR GIS data and data from La Selva biological research station of the Organization for Tropical Studies (OTS) was available (Table 5.2).

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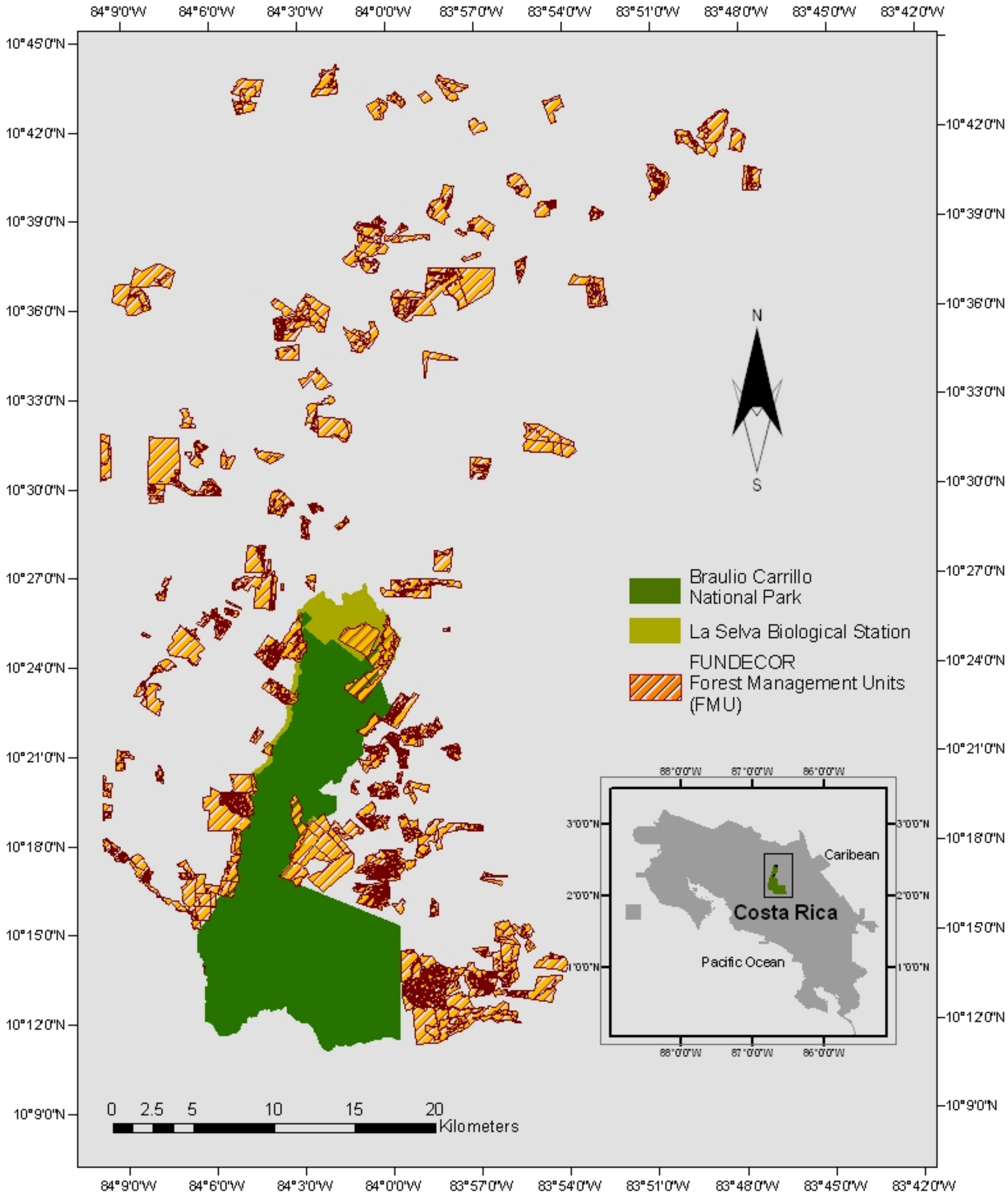

**Figure 5.1.** General location of study area (Data from FUNDECOR)

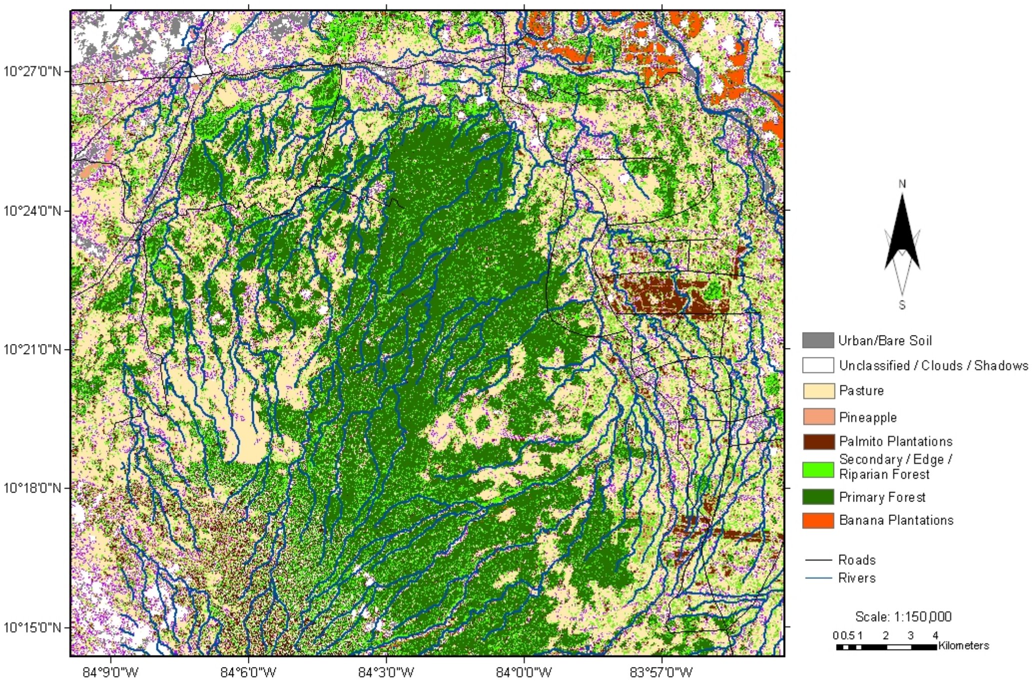

**Figure 5.2** Study area land cover-land use 2002. (source: Organization for Tropical Studies (OTS) and Fundacion para el Desarrollo de la Cordillera Volcanica Central (FUNDECOR)

**Figure 5.3.** a) Primary forest creek inside La Selva OTS biological research station, b) and c) pasture areas established after forest clearing. (Photos: Thomas Perea Bruckner, Costa Rica, June 2005).

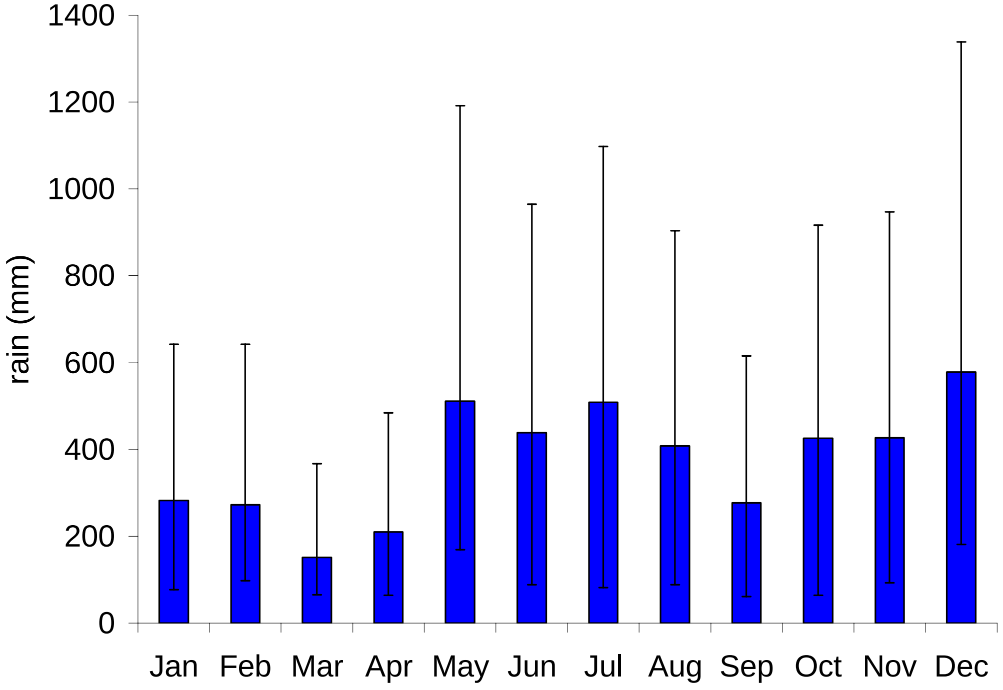

**Figure 5.4.** Ten year average (1995-2005) monthly rainfall data from Puente weather station located at La Selva (OTS) tropical forest biological research station, Puerto Viejo de Sarapiqui, Heredia, Costa Rica. (<www.ots.ac.cr/en/laselva/metereological.shtml>)

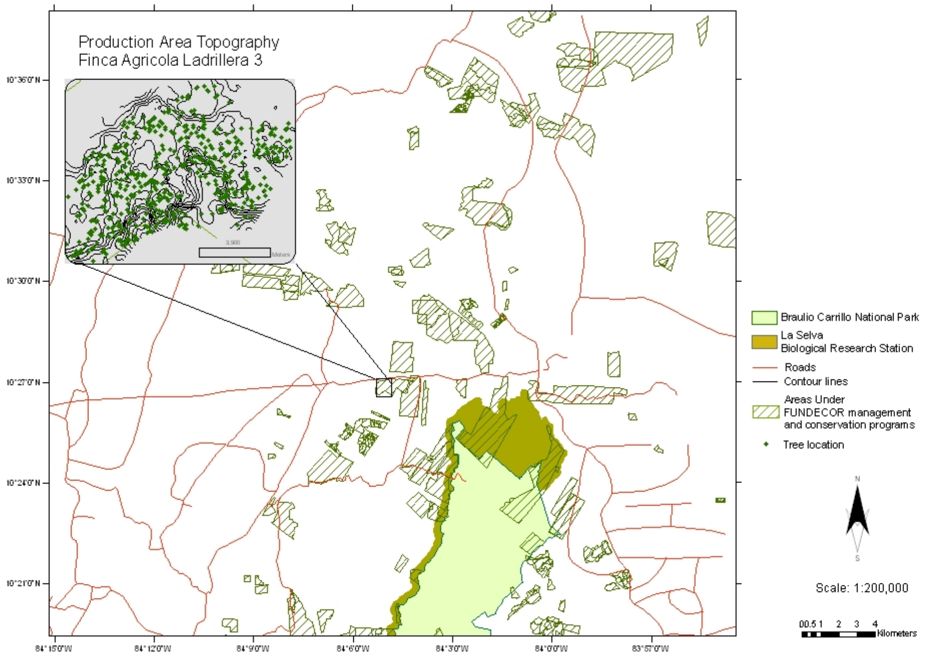

**Figure 5.5.** Forest Management Unit topography and tree locations **(**Data provided by Organization fro Tropical Studies [OTS] and Fundacion para el Desarrollo de la Cordillera Volcanica Central [FUNDECOR]).

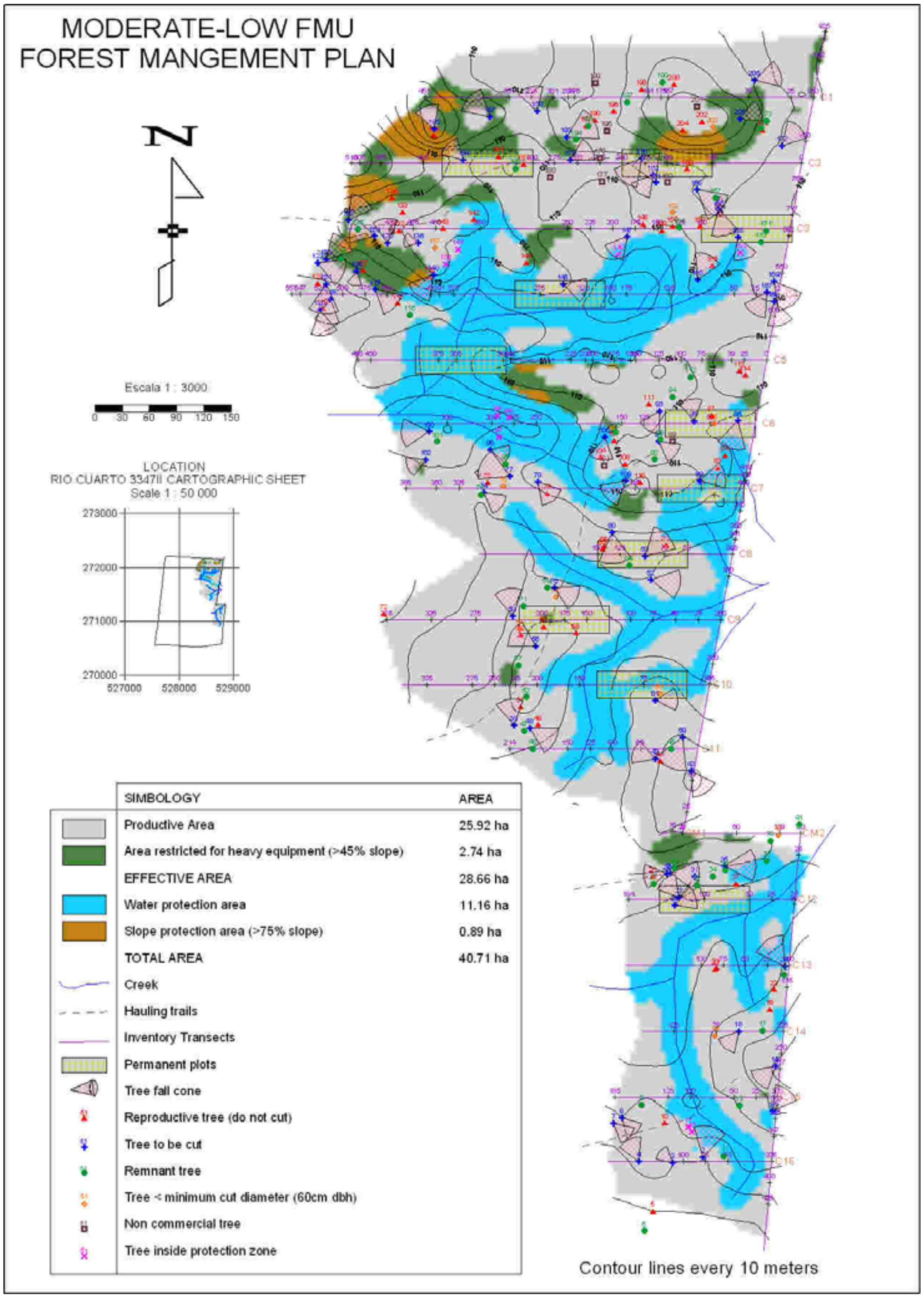

**Figure 5.6.** Example of forest management plan map. This map is used in the field by logging and monitoring crews. It allows easy navigation and application of management practices established based on FMU specific conditions. This specific area was sampled in the field (see Moderate-Low Forest in Chapter 8) (Source: FUNDECOR).

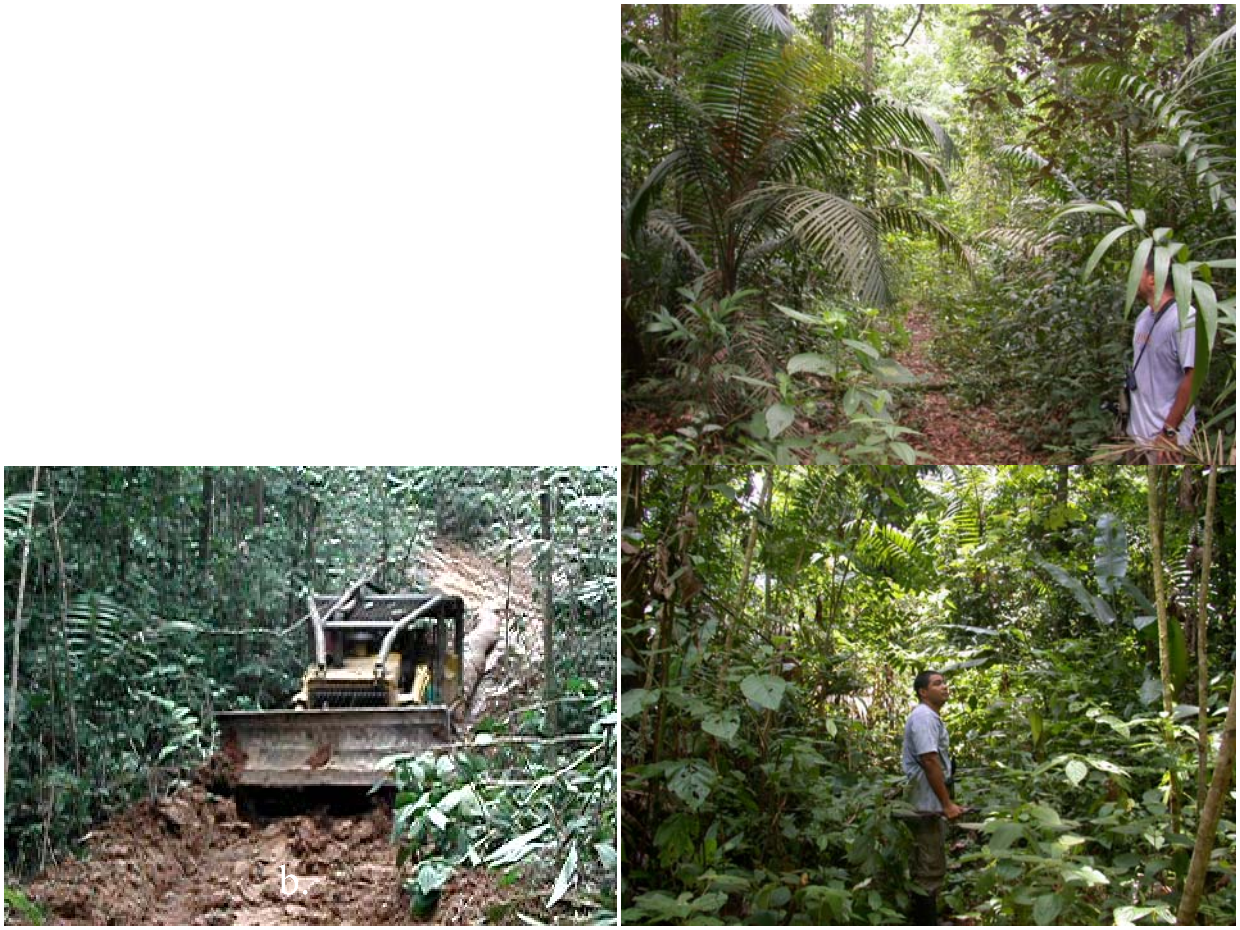

**Figure 5.7.** Hauling trails regeneration: **a**. during logging, **b.** 3 years after logging, and **c**. 6 years after logging. (Photos: Thomas Perea Bruckner, Costa Rica, June 2005).

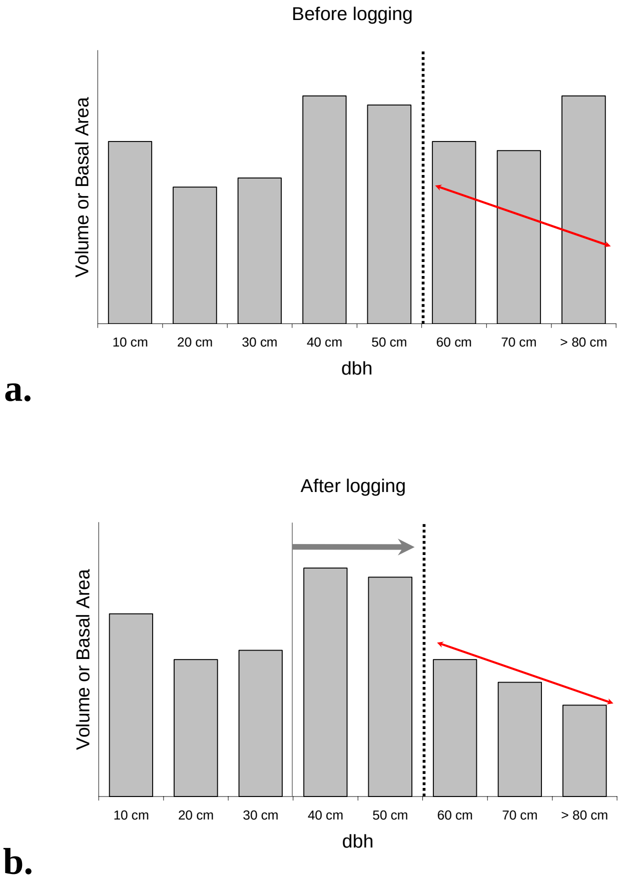

**Figure 5.8.** FMU forest cut cycle length definition. Part a. shows the typical diametric distribution (diameter at breast height categories) of commercial species in an unlogged area. Volume concentrations are around 40-50 cm dbh and above 80 cm. When a minimum cut diameter is established at 60 cm, as is the case in Costa Rica, logging reduces the volume above that threshold (red arrow). Sixty percent of volume above 60cm is usually logged. The objective, in order to insure a sustainable production, is to wait enough time before logging again so the volume concentrated around 40-50 cm reaches the 60 cm limit. Based on an average 0.5cm/year diameter increase for the species in the study area, a cut cycle length of 20 to 40 years needs to be established (FUNDECOR 2006).

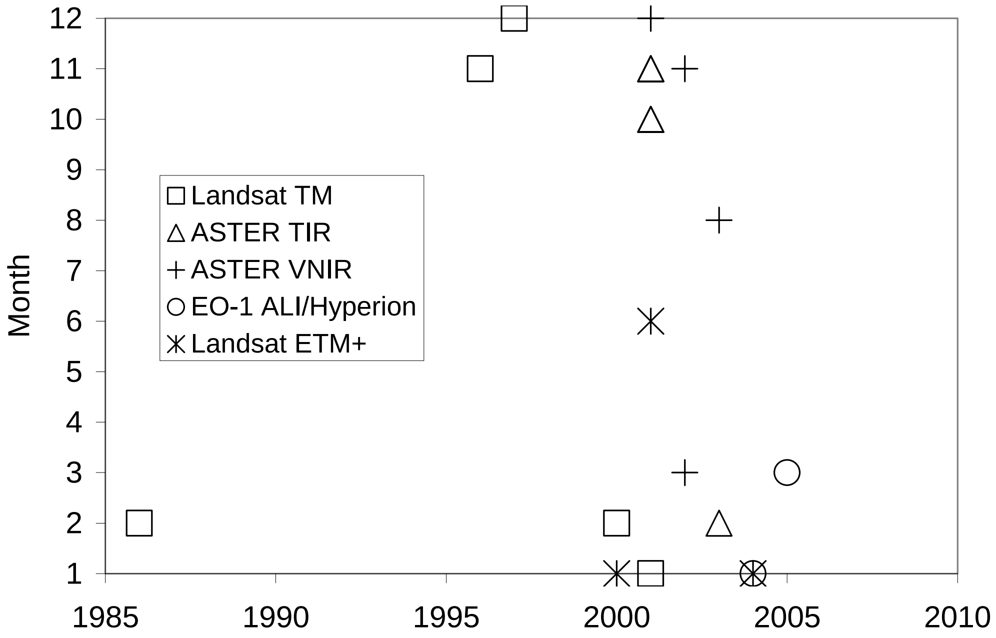

**Figure 5.9.** Available imagery with low cloud cover for the study area

**Table 5.1.** Costa Rica Forest Statistics

| | | |
|---|---|---|
| Forest Cover | Total forest area | 2,391,000 ha |
| | % of land area | 46.8% |
| | Primary forest cover | 180,000 ha |
| | % of land area | 3.5% |
| | % total forest area | 7.5% |
| Deforestation Rates, 2000-2005 | Annual change in forest cover | 3,000 ha |
| | Annual deforestation rate | 0.1% |
| | Change in deforestation. rate since '90s | -117.2% |
| | Total forest loss since 1990 | -173,000 ha |
| | Total forest loss since 1990 | -6.7% |
| Primary or "Old-growth" forests | Change in deforestation rate since '90s | -100.0% |
| | Primary forest loss since 1990 | -29.4% |
| Forest Classification | Public | 24.3% |
| | Private | 75.7% |
| Use | Production | 0.1% |
| | Protection | 1.9% |
| | Conservation | 24.5% |
| | Multiple purpose | 73.5% |
| Forest Area Breakdown | Total area | 2,391,000 ha |
| | Primary | 180,000 ha |
| | Modified natural | 1,319,000 ha |
| | Semi-natural | 888,000 ha |
| | Production plantation | 1,000 ha |
| | Production plantation: | 3,000 ha |
| Plantations | Plantations, 2005 | 4,000 ha |
| | % of total forest cover | 0.2% |
| | Annual change rate (00-05): | 200,000 ha |
| Number of tree species in IUCN red list | Number of native tree species | 117 |
| | Critically endangered | 4 |
| | Endangered | 33 |
| | Vulnerable | 74 |

| | | |
|---|---|---|
| Wood removal 2005 | Industrial roundwood | 1,932,000 m$^3$. |
| | Wood fuel | 468,000 m$^3$ |
| Value of forest products, 2005 | Industrial roundwood | \$122,122,000 |

Sources: Mckenzi (2004); FAO (2005); FAO/FRA (2000).

**Table 5.2.** GIS data available for study area

| Regional scale | | Forest Management Unit (FMU) scale | |
|---|---|---|---|
| Layer | Source | Layer | Source |
| Land use / land cover 2002, 2000 | La Selva (OTS)*, FUNDECOR | Contour lines | FUNDECOR-La Selva (OTS) |
| STRM 30 arcsec Digital Elevation Model | USGS** | Hauling Roads | FUNDECOR |
| Roads | FUNDECOR | Census Trails | FUNDECOR |
| Rivers | FUNDECOR | Permanent Plots | FUNDECOR |
| FUNDECOR FMU location | FUNDECOR | Tree locations | FUNDECOR |
| Urban Areas | FUNDECOR | Hydrology | FUNDECOR |
| Land use/cover change 1992-2000 | FUNDECOR | | |

* <http://www.ots.ac.cr/en/laselva/gis.shtml>

** http://srtm.usgs.gov/

# Chapter 6: Monitoring tropical forest spatio-temporal heterogeneity using the Wide Dynamic Range Vegetation Index (WDRVI)[9]

## *Abstract*

Sustainable management of tropical forests has been identified as one of the main objectives for conservation and management of carbon stocks. Thus, managers need tools to assess whether current management practices are sustainable. While sets of criteria and indicators have been developed to help managers, there is need to assess these indicators from an operational perspective. We present an approach using geospatial analysis to assess a key ecoindicator: 'The change in diversity of habitats as a result of human interventions is maintained within critical limits as defined by natural variation and/or regional conservation objectives'. Applying variography to the Wide Dynamic Range Vegetation Index (WDRVI) data from Landsat 5 TM imagery and comparing the changes of spatial structure before and after selective logging, we identified which managed forest areas exhibited significant differences with respect to natural reference areas.

## *Introduction*

Sustainable management of tropical forests has been identified as a main objective for global conservation of biodiversity and carbon stocks (CIFOR 2000). However, forest managers lack tools to evaluate the sustainability of their practices. Several sets of criteria and indicators (C&I) have been developed for managers to use as tools in the evaluation of the sustainability of their specific operations (Holvoet and Muys 2004). The majority of the indicators are based on basic scientific research in forest ecology with little regard for logistical considerations (Ghazoul 2001).

A tropical forest is not a steady-state ecosystem; each possesses its own disturbance history and regional dynamics that can create a mosaic of habitats differing in microclimate, structure, and composition (Morgan et al. 1994). The spatial distribution of regeneration stages produces the vertical and horizontal heterogeneities encountered in each forest area (Weishampel

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[9] Paper submitted to *International Journal of Remote Sensing*

et al. 2001) and thus forest management can change forest structural heterogeneity (Pinar and Putz 1996). Accordingly, the Centre for International Forestry Research established indicator I.2.1.2 to facilitate identification of potentially disruptive management: 'The change in diversity of habitats as a result of human interventions are maintained within critical limits as defined by natural variation and/or regional conservation objectives' (CIFOR 2000). Forest heterogeneity is usually surveyed in the field (Finegan et al. 2004), making it costly and limited in scope. With increasing area under management, field surveying becomes progressively less practical. An alternative approach is to use remotely sensed data (Lambin 1999; Lu et al. 2004).

To explore evaluating indicator I.2.1.2 using remote sensing, we characterized the spatio-temporal variability of the Wide Dynamic Range Vegetation Index (WDRVI; Gitelson 2004) obtained from two Landsat TM scenes (1986 and 2001) in managed and natural forest areas in Costa Rica.

*Data and Study area*

Reflectance Reflectance data from six natural and fifteen managed forest areas were obtained from two Landsat 5 TM images (WRS-2 Path 15, Row 53) from 06 February 1986 and 14 January 2001. Managed forest areas correspond to forest management units (FMUs), selectively logged (≤ 5 trees/ha) between 1998 and 2000. All forests are located in the canton of Sarapiqui in Costa Rica, between 100 and 1500 meters above sea level, near Braulio Carrillo National Park. Natural reference areas were selected inside Braulio Carrillo National Park and La Selva Biological Station.

Analysis of forest structural heterogeneity was made using the Wide Dynamic Range Vegetation Index (Gitelson 2004):

$$WDRVI = \frac{\alpha * \rho_{NIR} - \rho_{RED}}{\alpha * \rho_{NIR} + \rho_{RED}} \qquad [1].$$

The coefficient α can be understood as a tuning knob that nonlinearly adjusts the gain on the index: when α equals 1, the WDRVI is the NDVI; when α equals $(\rho_{RED} / \rho_{NIR})$, the WDRVI equals zero. It can enhance the dynamic range of the NDVI when the leaf area index (LAI) exceeds about two, as is commonly encountered in tropical forests (Gitelson 2004; Viña et al. 2004).

*Methods*

Following Henebry et al. (2004), different α values were used to evaluate the effect on observed WDRVI dynamic range within the study area: the larger the range, the greater the bandwidth. Thus, we used α to 'tune' WDRVIs dynamic range to the specific conditions of the scenes to enhance observed spatial heterogeneity.

The spatial variation of the WDRVI values occurring in each forest area was described by means of variography (Goodin et al. 2004). We chose omnidirectional variograms as appropriate neutral models of spatial variation. In every case we fit a spherical model [2] because it almost always provided the better fit:

$$\gamma(h_i) = \begin{cases} \gamma(h_0) + \left[ Sill * 1.5\left(\frac{h}{a}\right) - 0.5\left(\frac{h}{a}\right)^3 \right] & if\ h \le a \\ \gamma(h_0) + Sill & otherwise \end{cases} \qquad [2]$$

where $h_i$ corresponds to the distance in the specific $i^{th}$ lag class, $a$ corresponds to the range distance, and *Sill* corresponds to the value of the semivariance when the range distance has been reached, and $\gamma(h_0)$ corresponds to the nugget variance at lag $h$=0 (Goovaerts 1997).

The parameter coefficients of the spherical model (range, sill, and nugget variance) provide a representation of the structure of the heterogeneity within each study area. The range denotes the spatial dependence of the data. The sill indicates the observable spatial heterogeneity. The nugget describes the intrinsic variability in the data that may occur at scales finer than sampled and/or measurement errors (Atkinson and Tate 2000). The sum of the sill and the nugget is the total observed variation $\gamma(h)$ at $h > a$. The total variation and range together summarize the spatial heterogeneity of WDRVI values.

We used VARIOWIN 2.21 (Panatier 1993-1998) for variography and model fitting. We inferred changes in forest structure heterogeneity between 1986 and 2001 from differences in model parameter coefficients compared using pairwise nonparametric tests (Conover 1998).

*Results*

Analysis of the effect of α on the observed range of the WDRVI revealed that values between 0.25 and 0.35 yielded the largest dynamic ranges in all the forest areas (Figure 6.1). Accordingly, we used an average α value of 0.30 to calculate the WDRVI. The resulting histograms of the WDRVI values were significantly wider than those of the NDVI (Figure 6.2). The observed variability of the WDRVI in both periods was substantially greater than that of the NDVI (1986: $CV_{NDVI}$=7.30% vs. $CV_{WDRVI}$=51.4%; and 2001: $CV_{NDVI}$=7.20% vs. $CV_{WDRVI}$=50.8%). The variance of the WDRVI was significantly larger than that of the NDVI (Kruskal-Wallis test p-value <0.001).

Observed changes in WDRVI total variation and range between 1986 and 2001 in natural reference areas were not significant (Sign test: $\text{p-value}_{\text{total variation}}$ >0.999; $\text{p-value}_{\text{range}}$ = 0.687) (Figure 6.3). No significant change in spatial heterogeneity was expected in these areas, since there had been no documented changes in the natural disturbance regime between 1986 and 2001.

Managed areas showed significant changes in total variation between 1986 and 2001 (Sign test: p-value = 0.03; Figure 6.3a). However, no significant differences were found either in the ranges (Sign test p-value = 0.3; Figure 6.3b) or in the mean values between natural and managed areas (Mann-Whitney test $\text{p-value}_{\text{total variation}}$ = 0.12; $\text{p-value}_{\text{range}}$ = 0.64).

### *Discussion*

No significant differences were observed in average total variation or range between natural reference and managed areas; thus, we conclude that the management scheme for these FMUs considered as a whole has not generated significant changes in forest structural heterogeneity. This finding was not unexpected as these forests are the object of low impact logging under FSC standards, which aim to generate the lowest possible alteration from natural dynamics (Finegan et al. 2004). However, this aggregate result provides little insight into the particular pattern manifested by each FMU.

A complementary perspective uses the observed changes in the reference forests as a dynamic baseline by constructing 99% confidence intervals based on the linear regression of later (2001) values of total variation and range in natural reference area on earlier (1986) values (Figure 6.3). Not every managed area is located within the confidence intervals enabling quick identification of managed areas manifesting changes outside the inferred limits of natural variation. From this perspective, 7 out of 15 FMU were beyond the 99% confidence envelope for both total variation and range (Figure 6.3). Using these results, managers can develop a triage for biodiversity field surveying (Aguilar-Amuchastegui and Henebry 2006)

Variography applied to the WDRVI enabled characterization of the structural heterogeneity of dense tropical forests and its change through time. As expected, based on prior literature, selectively logged forests showed larger changes in spatial structure than the natural ones. A key step forward here was the use of nearby forest areas under a natural disturbance regime to provide a dynamic baseline to establish the limits of natural variation.

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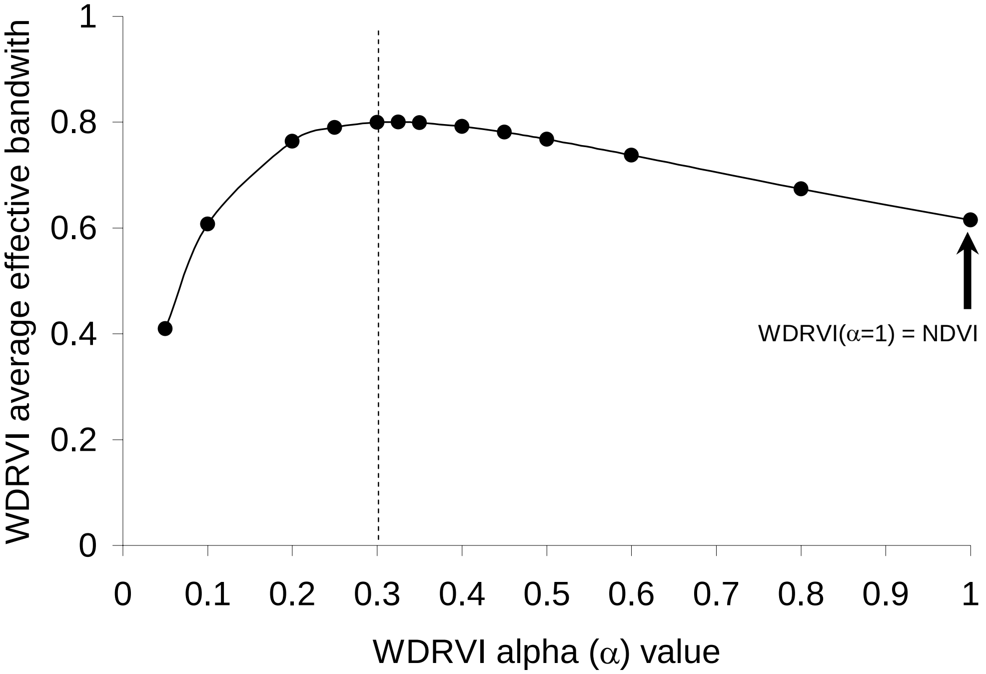

**Figure 6.1.** Histogram range (or effective bandwidth) observed in WDRVI values as a function of α values.

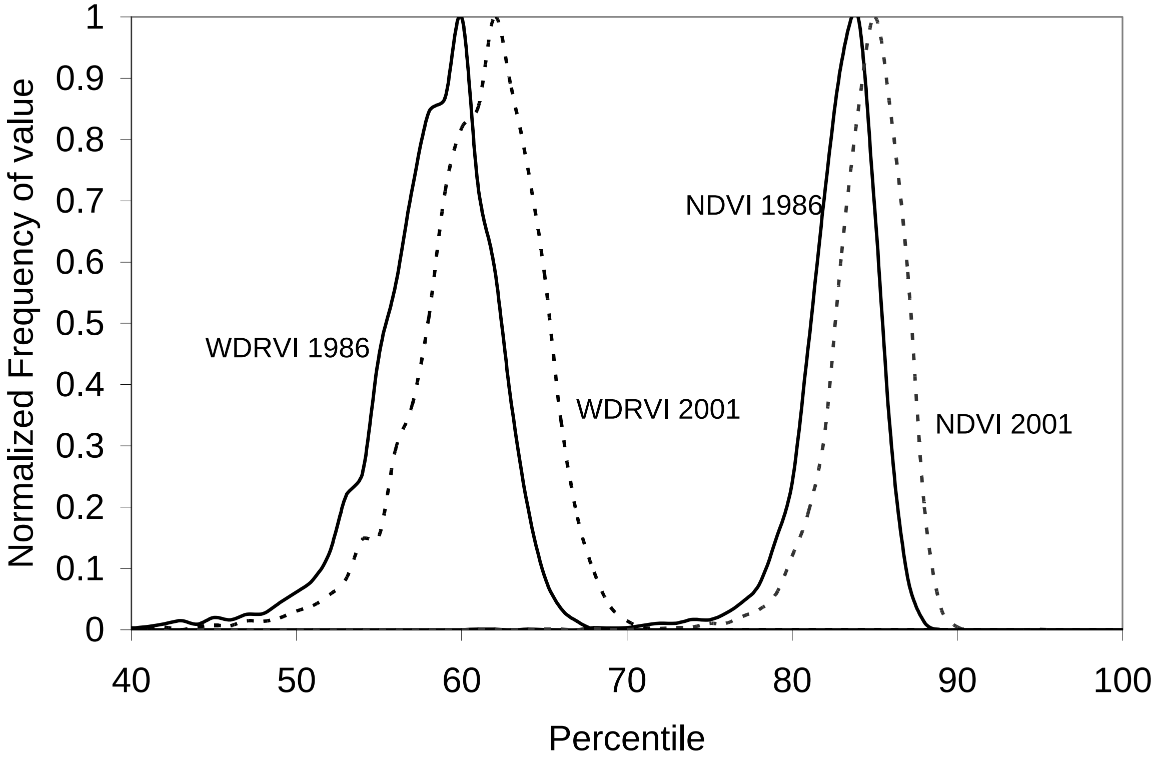

**Figure 6.2.** Normalized histograms for the values of NDVI and the WDRVI ($\alpha$=0.3) for all study areas in 1986 (solid line) and 2001 (dotted line).

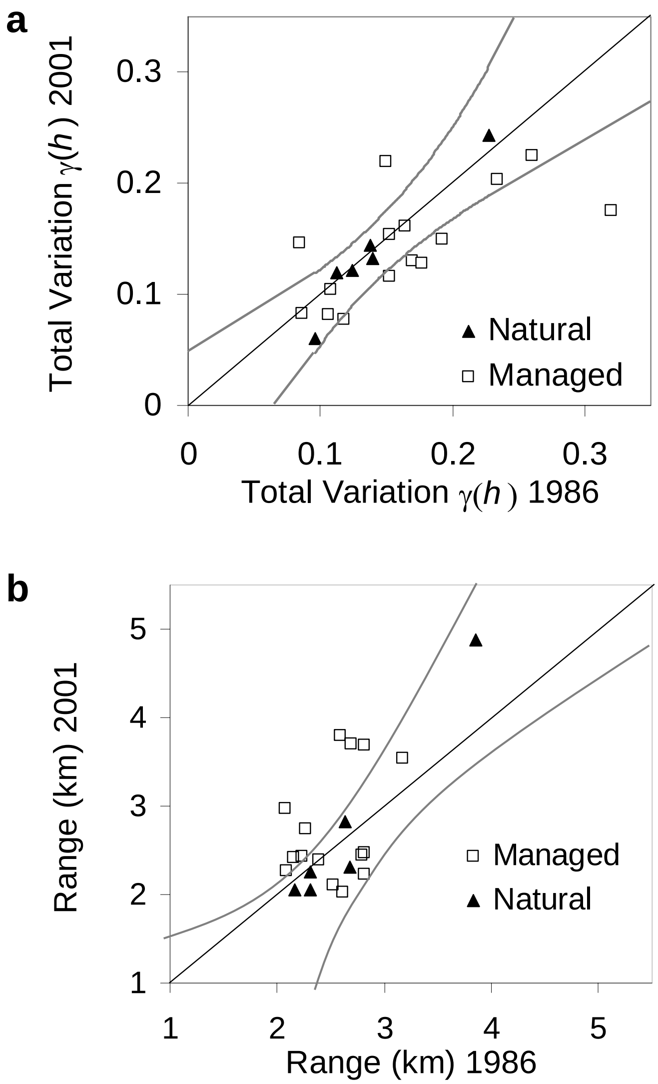

**Figure 6.3.** Temporal correspondence between the total variation and range as estimated with the WDRVI (α=0.3) in each area between 1986 and 2001. Confidence intervals (99%) allow identification of managed areas that show changes outside of natural variation. Confidence intervals are based on linear regression of model coefficients for the natural reference areas between periods (Adjusted $r^2$=0.94 for total variation and range with p-value$_{\text{total variation}}$ = 0.001 and p-value$_{\text{range}}$ < 0.001).

# Chapter 7: Monitoring sustainability in tropical forests: How changes in canopy spatial pattern can indicate forest stands for biodiversity surveys[10]

*Abstract*

Sustainable management of tropical forests has been identified as one of the main objectives for global conservation and management of carbon stocks. Toward this goal, managers need tools to determine whether current management practices are sustainable. Several international initiatives have been undertaken for the development of criteria and indicators to aid managers in moving towards sustainable practices. Despite these efforts, the question of how to apply and assess indicators remains to be answered from an operational, field-based perspective. Field surveys are expensive and time-consuming when management areas are large and in the face of logistical constraints. Thus, there is a need for an approach to prioritization. We sought to determine whether satellite imagery can be used, in conjunction with standard forest management data, to identify and rank priority areas for field surveys of bioindicators. The study area in Costa Rica, in forest areas managed by the Fundacion para el Desarrollo de la Cordillera Volcanica Central (FUNDECOR), was imaged by Landsat 5 TM in 1986 and 2001. Through spatial statistical analysis applied to the Wide Dynamic Range Vegetation Index, we were able to quantify and rank changes in canopy spatial structure. The resulting categories can be used by forest managers to identify which areas are in need of field surveys. More generally, we show how to generate a moving baseline for change analysis and evaluate for significant deviations in spatial structure.

*Introduction*

Sustainable management of tropical forests has been identified as a key objective for global conservation, as they are among the largest and most endangered biomes (CIFOR 2000). Forests that are neither economically productive nor protected by conservation status are at risk

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[10] Paper published in IEEE *Geophysical Research Letters*, July 2006

of transformation into other land uses. Not all forest management practices are sustainable, but managers lack tools to gage their sustainability.

Several initiatives have been undertaken to create sets of criteria and indicators (C&I processes) to be used as tools in the evaluation of the sustainability of specific operations within a given timeframe and with the ability to monitor sustainability trends (CIFOR 2000; McGinley and Finegan 2003; Finegan et al. 2004; Ghazoul and Hellier 2000; Ghazoul 2001). These initiatives consider a wide variety of indicators that managers can assess in order to build an integrated view of the sustainability of their management practices. The majority of the indicators are based on scientific and theoretical approaches (CIFOR 2000; Ghazoul 2001).

In 1994, the Center for International Forestry Research (CIFOR) initiated one of the leading C&I processes. CIFOR brought together teams of experts on the ecological, economic and productive aspects that affect forest management sustainability, to design specific sets of C&I for each area, based on research, conceptual framework and field evaluations. The process resulted in a generic template that forest managers can use and adapt to the specific conditions of their operations (CIFOR 2000; Ghazoul 2001). They stated that biodiversity maintenance is a surrogate of ecological sustainability; thus, many sustainability indicators seek to assess biodiversity levels within managed areas. However, the question of how to implement these indicators remains to be answered from an operational viewpoint, where logistical constraints become critical.

Tropical forests are not steady-state ecosystems; they exhibit a range of natural disturbance levels (Ghazoul and Hellier 2000). Their historical range of variability (Morgan et al. 1994) embraces a dynamic of disturbances in addition to gap dynamics and creates a mosaic of habitats differing in microclimate, vegetation structure, and faunal composition. The distribution of these patches generates the specific vertical and horizontal heterogeneity of a forest. Management practices can change forest heterogeneity, depending on its harvest intensity (as measured by the number of trees or the basal area or the cubic meters of wood removed per ha) (Ghazoul and Hellier 2000; Morgan et al. 1994; Delgado et al. 1997). Accordingly, CIFOR (2000), established as one of its indicators *"The change in diversity of habitats as a result of human interventions are maintained within critical limits as defined by natural variation and/or regional conservation objectives"*. Thus, a management scheme may be considered sustainable if it maintains the relative abundance and distribution of the successional stages that provide forests its diversity of habitats within the limits of natural variability. In contrast, forests exhibiting significant differences either in comparison with natural control areas or before and after harvesting (surveyed following a reasonable recovery period) would constitute *prima facie* evidence of unsustainable forest management.

Forest heterogeneity (diversity of habitats) and other indicators are usually surveyed in the field, which makes them very limited in extent and time-consuming (Finegan et al. 2004; Ghazoul 2001; Aguilar-Amuchastegui 1999). Therefore, as the number of forests to be surveyed or the total area under management increases, the personnel and time required increases and comprehensive surveying becomes impractical.

On the other hand, for the same reasons, surveys generally only compare natural and managed areas directly —as a surrogate of before and after harvest—without considering the intrinsic dynamics of these natural forests (CIFOR 2000; Finegan et al. 2004; Aguilar-Amuchastegui 1999). Such an approach can confound natural variation with the effects of management practices (Tuomisto et al 1995). Furthermore, the approach provides only a binary characterization of managed areas as similar or dissimilar to natural areas without addressing the magnitude of the dissimilarity or its statistical significance (Fahrig 2003).

Forest spatial pattern may be detected and measured remotely by means of spaceborne sensors (Weishampel et al. 1998; 2001; Wulder et al. 2004). Remotely sensed data can be used to estimate biophysical parameters of vegetation cover through the use of vegetation indices (Baret and Guyot 1991; Gitelson et al. 2003; Seto et al. 2004). Given earth observation data spanning more than three decades and multiple sensors, there is the potential to characterize dynamic baselines in tropical forests that can embrace intrinsic processes of disturbance and regeneration in natural areas. Here, we have used a pair of Landsat 5 TM images to demonstrate how to characterize spatio-temporal variation in canopy structure and compare managed and natural areas.

## *Methods*

### *Vegetation Indices*

The analysis of forest canopy spatial patterns was made through the study of the spatial variability of a new vegetation index related to the Normalized Difference Vegetation Index (NDVI) (Rouse et al. 1974): specifically, the Wide Dynamic Range Vegetation Index (WDRVI)

(Gitelson 2004; Henebry et al. 2004). The NDVI has been commonly used to relate to biophysical characteristics of vegetation such as leaf area index (LAI), fractional vegetation cover, or aboveground biomass (Gitelson 2004; Henebry et al. 2004; Viña et al. 2004). Yet, the NDVI begins to lose sensitivity when LAI is moderate (>2); thus, in high LAI environments—such as tropical forests—ecologically significant changes in canopy structure may not be detectable using NDVI (Henebry et al. 2004; Viña et al. 2004 ).

The WDRVI [7.1] is a generalization of the NDVI proposed for use with denser vegetation (Henebry et al. 2004; Viña et al. 2004 ):

$$WDRVI = \frac{(\alpha \times \rho_{NIR}) - \rho_{RED}}{(\alpha \times \rho_{NIR}) + \rho_{RED}} \qquad [7.1]$$

where $\rho_{\mathrm{NIR}}$ is near infrared reflectance (Band 4 of Landsat TM), $\rho_{\mathrm{RED}}$ is red reflectance (Band 3 of TM), and $\alpha$ is a weighting coefficient (Gitelson 2004). By down-weighting the contribution of the $\rho_{\mathrm{NIR}}$ with $\alpha < 1$, the value of $\alpha^*\rho_{\mathrm{NIR}}$ approaches $\rho_{\mathrm{RED}}$, thereby improving the sensitivity of WDRVI to changes in vegetation biophysical parameters (Henebry et al. 2004; Viña et al. 2004 ). Note that if $\alpha=1$, then WDRVI=NDVI.

Following the approach in (Henebry et al. 2004), we used [7.2] to determine $\alpha=0.3$ to be optimal with these data:

$$\alpha_{est} = 2 \times \frac{(average\, \rho_{RED})}{(Maximum\, \rho_{NIR})} \qquad [7.2]$$

*Study Area*

The study area is located in the canton of Sarapiqui, in the Atlantic slope of the Cordillera Volcanica Central in Costa Rica. Analysis focused on 15 forest management units (harvested 1998-2000) and 6 non-harvested natural forest areas managed by FUNDECOR[1]. Managed units

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[1] Fundacion para el Desarrollo de la Cordillera Volcanica Central (FUNDECOR) is a Costa Rican NGO that provides private landowners with forest management services under forest management certification standards established by the Forest Stewardship Council (FSC).

were selectively harvested at very low intensity (<5 trees/ha removed, minimum diameter at breast height is 60cm; see Table 7.1). Each management unit corresponds to a privately-owned forest area that is subject to a management plan that is tuned to its specific conditions (*e.g.,* topography, location, number of trees, species composition). The size of the forest management units ranged from 14 to 605 ha (Table 7.1).

### *Data*

Image data were two Landsat 5 TM scenes (WRS-2 Path 15, Row 53): one before harvest from February 1986 and the other after harvest from January 2001. Images were converted from DN to reflectance following (Markham and Baker 1986) and then coregistered. No atmospheric corrections were applied for several reasons: (1) elevations in the forest areas ranged from 100-1500 m above sea level; (2) no ground data were available; (3) we sought an approach that can be readily implemented by a forest manager's geospatial technician and it requires more specialized knowledge that is likely available to do atmospheric correction well; and (4) our analysis relied on the relative differences between pixels within a single image date and then comparing these derived relationships through time, thus atmospheric correction is less of a concern ( Henebry and Su 1993; Goodin et al. 2004).

### *Canopy Spatial Patterns*

We assessed canopy spatial patterns by fitting a spherical model to describe the semivariogram obtained for each forest management unit at each time point (He et al. 1996; Légendre and Fortín 1989; Légendre and Légendre 1998; Duteuil and Légendre 1993; Henebry 1993; Goodin and Henebry 1997). We used VARIOWIN 2.21 (Pannatier 1995, 1993-1998) for the variography and model fitting. Estimates of the three parameters of a spherical model (sill, nugget variance, and range) served to summarize the spatial pattern observed among WDRVI values. Change in spatial pattern between 1986 and 2001 was inferred from change in model parameter estimates.

Range refers to the distance where the variance no longer exhibits spatial dependence. The sill corresponds to the portion of the total variation that exhibits spatial dependence and the nugget variance corresponds to local variability occurring at scales finer than the sampling interval. The possible sources of this finer scale variability include instrument error, sampling error, or the intrinsic heterogeneity of the measured phenomenon (He et al. 1996; Légendre and Fortín 1989; Légendre and Légendre 1998; Duteuil and Légendre 1993; Henebry 1993; Goodin and Henebry 1997). Beyond the range, the sum of the sill and the nugget variance constitute the total variation.

We sought significant differences in vectors that have both magnitude and orientation information. The orientation information is on a periodic scale ($0\text{-}359^o$); thus, we need to use circular statistics to calculate correctly the means and variances (Finegan and Camacho 2000) of the changes between natural and managed forests. Changes observed in natural forests served as a baseline against which to assess the significance of the changes observed in managed forests.

### *Results and Interpretation*

Each forest exhibited change in spatial structure between acquisitions (Figure 7.1). Natural forests exhibited, in general, a reduction in the sill/total variation ratio accompanied with an increase in the nugget variance. Ranges did not change substantially, except in one case (Figure 7.1a). Managed forests also exhibited a general pattern of reduction in the sill/total variation ratio. However, in this case, some forests presented an increase in the ratio accompanied by a decrease in the nugget variance. Changes in range were typically greater than the ones observed in natural forests, but the direction of the change was not uniform: some forests increased and others decreased (Figure 7.1b.). Analysis of the mean change angle and magnitude between natural and managed forests revealed no significant differences between groups (Table 7.2).

Even though management can increase forest heterogeneity (CIFOR 2000; Finegan et al. 2004; Ghazoul and Hellier 2000; Delgado et al. 1997; Finegan and Delgado 2000; Finegan et al. 2001; Farthing 2001) comparisons of canopy spatial patterns in terms of the mean change angle and magnitude in sill/total variation – range relation (Figure 7.1 and Table 7.2) did not show significant differences between natural and managed forests. This result was expected since the current management of these forests aims at sustainability through very low intensity harvesting, under FSC standards, that specifically tries to generate the least possible changes from natural dynamics (Batschelet 1981).

However, comparison of group behaviors does not address specific changes within particular management units. To identify critical change thresholds for individual management units, it is important to consider that each forest has an intrinsic disturbance history as well as a disturbance context dictated by regional processes such as fragmentation and extreme meteorological events (Ghazoul 2001; Farthing 2001; Fahrig 2003). Ideally, each forest area would provide its own dynamic baseline from which to assess the impact of particular management practices (Ghazoul 2001; Farthing 2001).

Figure 7.2 summarizes the changes in observed canopy spatial structure for forest management unit on an individual basis. The elliptical areas represent the mean change in natural forests plus two and three standard deviations (SD). The 95% confidence ellipse (the inner ring) bounds the observed natural range of variation. This threshold could enable managers to identify those forests that exhibit anomalous changes in spatial structure. Many, but not all, managed forests fall outside of this 2SD threshold. However, if we add another standard deviation to the threshold to account for the effects of recent harvest (within a decade), then only one managed forest falls outside the 3SD limit.

The location of managed areas with respect to each of the boundaries can be used as a mean of ranking forests for field surveys of sustainability bioindicators. For example, in the case of the solitary outlier beyond 3SD, managers might rank it as the highest priority for surveying, followed by the seven units with change rates between 2SD and 3SD, with the rest within the 2SD boundary ranking as lower priority (Table 7.1).

It may be expected that observed changes in forest spatial structure and canopy heterogeneity would be directly related with harvest intensity. However, our results show no significant correlation between the mean change magnitudes or angles in managed forests and their harvest intensities (Table 7.2). Further, we found no significant differences between priority rankings shown in Table 7.1 and harvest intensities (ANOVA p-value = 0.375). These findings reinforce the notion that change analysis in forests need to be conducted individually as management units. Every managed forest has a particular disturbance history and environmental context which leads to high variability within the group of managed forests. While this makes group generalizations difficult, managers do not operate on groups of forests, but rather specify treatments for individual units. This mismatch between the scientific urge to generalize and the managerial requirement to specify underlines the need for the kind of triage approach.

### *Conclusions*

We have presented an exploratory study of how dynamic baselines can be extracted from a pair of scenes. We have presented a first approach to using Landsat–scale imagery for prioritizing forests areas for field surveys of bioindicators. The next step requires field verification of the rankings. If bioindicators of ecological sustainability change similarly to canopy spatial patterns, then this approach could be extended from sustainable forest management to biodiversity conservation.

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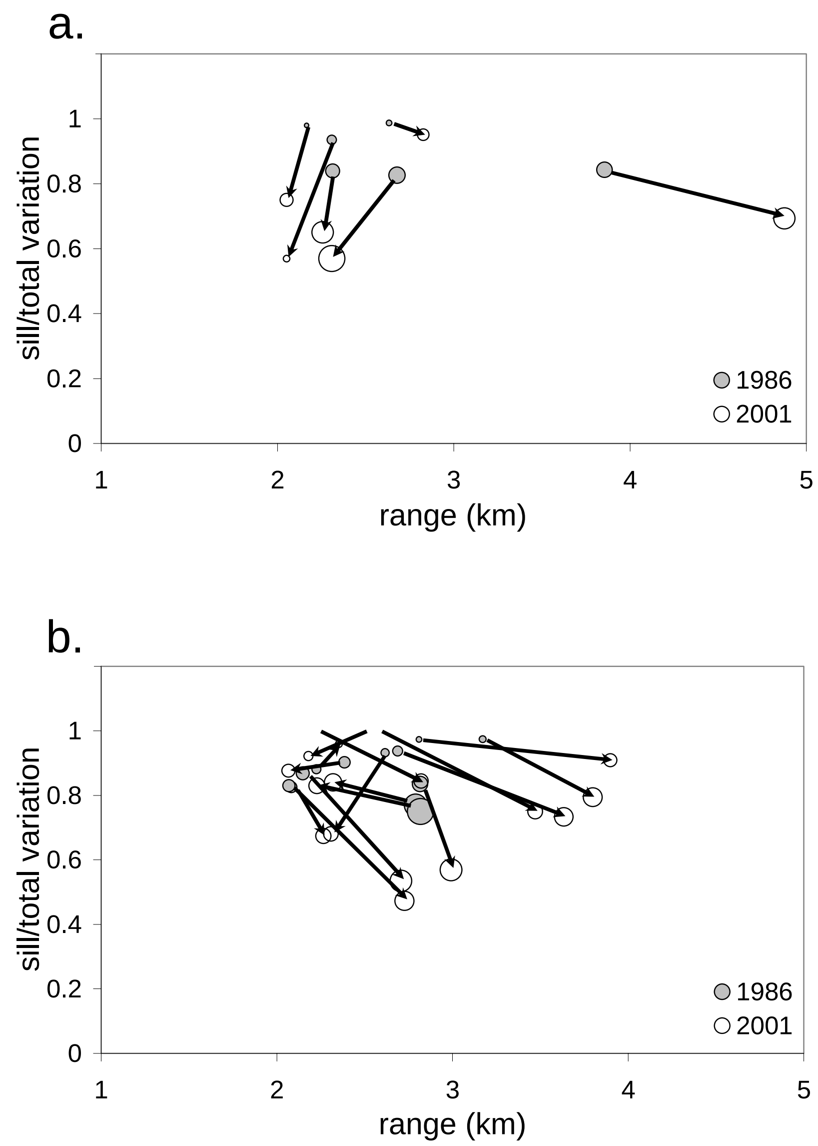

**Figure 7.1.** Changes in the relation sill/total variation - range observed in natural forests (a) and managed forests (b) WDRVI α=0.3 values between 1986 and 2001. Bubble size represents size of the nugget variance, which was sometimes equal to zero. Arrows connect the same forests between periods.

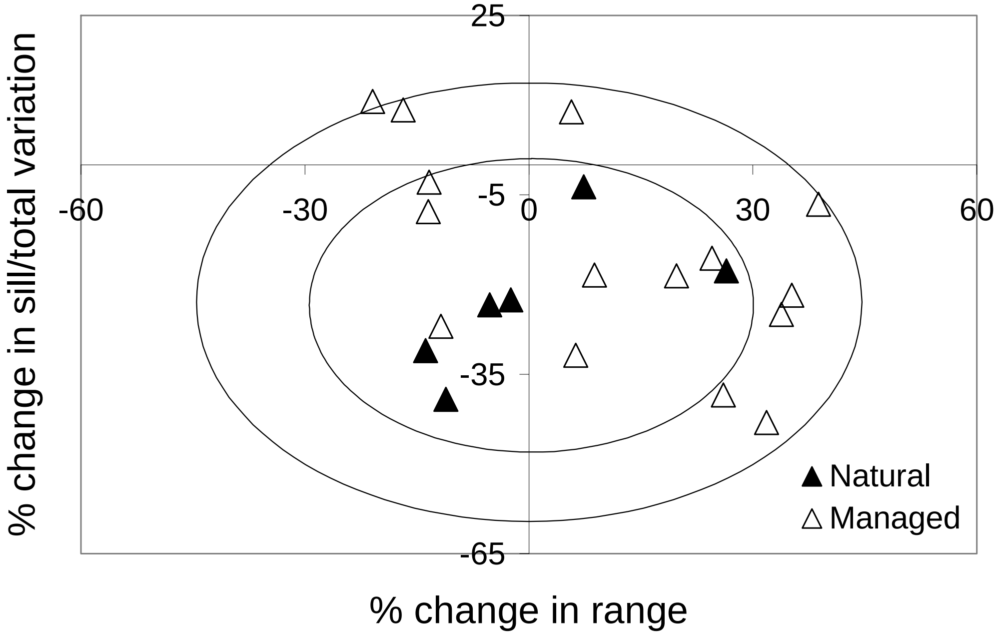

**Figure 7.2.** Rates of change (%) in spatial structure as observed in natural and managed forests between 1986 and 2001 using WDRVI with α=0.3. The elliptical boundaries show the average changes in natural forests plus two and three standard deviations. Location of a specific forest with respect to these thresholds enables an interpretation in terms of deviation from observed natural rates of change.

**Table 7.1.** Field Survey Priority Level. (Based on WDRVI α=0.3 values spatial pattern change rates).

| Forest | Year of harvest | Area (ha) | Trees cut | Harvest intensity (trees/ha) | Priority level |
|---|---|---|---|---|---|
| 1 | 1998 | 62 | 114 | 1.85 | Low |
| 2 | | 22 | 100 | 4.55 | Intermediate |
| 3 | | 118 | 231 | 1.96 | Intermediate |
| 4 | | 40 | 73 | 1.83 | Intermediate |
| 5 | | 65 | 88 | 1.35 | Intermediate |
| 6 | | 45 | 85 | 1.89 | Low |
| 7 | | 72 | 145 | 2.01 | Low |
| 8 | | 357 | 1108 | 3.10 | Intermediate |
| 9 | 1999 | 14 | 41 | 2.93 | Low |
| 10 | | 27 | 45 | 1.67 | Low |
| 11 | | 50 | 96 | 1.92 | High |
| 12 | 2000 | 605 | 616 | 1.02 | Low |
| 13 | | 58 | 62 | 1.07 | Low |
| 14 | | 137 | 647 | 4.72 | Intermediate |
| 15 | | 145 | 160 | 1.10 | Intermediate |

**Table 7.2.** Differences in Change from 1986 to 2001 between Natural and Managed Forests

| Change dimension | Angle | Magnitude |
|---|---:|---:|
| Test | Watson U2 test | ANOVA (F-test) |
| P-value | 0.50 > x > 0.20 | 0.40 |

# Chapter 8: Indicators for tropical forests: forest structure dynamic change, logging intensity, and dung beetles[11]

## *Abstract*

Sustainable management of tropical forests has been identified as one of the main objectives for conservation of global biodiversity and management of carbon stocks. To achieve this goal, managers need tools to assess the sustainability of current management practices. Several international initiatives have undertaken the development of sets of criteria and indicators to help managers move towards sustainability. Among the indicators considered, the structure and composition of d*ung beetle communities have been identified as excellent indicators of ecological sustainability. However, as occurs with most indicators of the ecological sustainability of forest management, dung beetle surveys require intensive field work making their application over large areas expensive, time consuming, and logistically challenging. A need for prioritization is evident. This works presents a novel* approach to the assessment of the Center for International Forestry Research (CIFOR) ecological sustainability indicator I.2.1.2: "The change in diversity of habitats as a result of human interventions is maintained within critical limits as defined by natural variation and/or regional conservation objectives". Using by variography of vegetation index data derived from remote sensors, we show how the differences observed between forest management units and natural areas forest structure heterogeneity can be used to identify priority areas for field survey of ecological sustainability indicators and how these priorities are reflected in dung beetles community structure and composition. The link between temporal change in forest structural heterogeneity, logging intensity, and dung beetle community structure and composition is established by means of correlation analysis and matrix regression modeling. A logging intensity threshold of 4 trees per hectare is identified as the transition to significant differences in forest spatio-temporal heterogeneity and the richness and diversity of associated dung beetle communities.

## *Introduction*

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[11] paper submitted to *Ecological Applications*

Sustainable management of tropical forests has been identified as a main objective for global conservation of biodiversity and carbon stocks as they are among the most diverse and endangered biomes (CIFOR 2000; FAO 2005; Vieira et al. 2004). Any forest not considered economically productive or protected by conservation status is at risk of conversion into other types of land cover. However, not all forest management schemes are sustainable and managers lack tools to evaluate the sustainability of their practices. Based on this deficiency, several initiatives have undertaken the development of sets of criteria (a standard that a management is judged by [CIFOR 2000]), indicators (any variable or component of the forest ecosystem or management system used to infer the status of a particular criterion [CIFOR 2000]) (C&I processes) that managers can use as tools in the evaluation of the sustainability of their specific operations (CIFOR 2000; Ghazoul and Hellier 2000; Franc et al. 2001; McGinley & Finegan 2003; Finegan et al. 2004). The majority of the indicators proposed are based on scientific research for conservation of biological diversity in managed forest systems (Stork et al. 1997; CIFOR 2000).

C&I processes have become very popular worldwide. For example, Holvoet and Muys (2004) were able to classify 164 different C&I standards under development according to their level of application: nation, region, and forest management unit (FMU); geographic origin; and biophysical and socio-economic aspects. The scope of the standards in developing countries has emphasized the social and economic aspects of sustainability; in contrast, developed countries have emphasized ecological function. In most cases, the question of how to apply and assess these C&I remains to be answered from an operational, field-based perspective (Ghazoul and Hellier 2000; Franc et al. 2001; McGinley & Finegan 2003; Finegan et al. 2004).

Dung beetles are commonly proposed as indicators of biodiversity due to their close relationship with all types of vertebrate fauna dung and their role as decomposers (Halffter and Favila 1993; Hill 1996; Favila and Halffter 1997; Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000; Davis et al. 2001; Halffter and Arellano 2002; Pineda et al. 2005; Scheffler 2005; Andresen 2005). They play important roles in forest regeneration, e.g., seed dispersal and nutrient cycling (Hanski and Cambefort 1991; Halfter and Favila 1993; Favila and Halftter 1997; Aguilar-Amuchastegui 1999, Aguilar-Amuchastegui et al. 2000; Davis et al. 2001; Andresen 2002). Dung beetles have proven to be an effective indicator group that can be used by forest managers and workers to survey ecological sustainability indicators such as the CIFOR

(Center for International Forestry Research [2000]) indicator “The richness/diversity of selected groups shows no significant change” (I.2.1.4.) and verifiers “Species richness of prominent groups is maintained or enhanced” (V.2.1.4.1) and “Temporal change in species richness is not significant” (V.2.1.4.6.) (Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000). They also have been proposed for forests management certification surveys (Finegan et al. 2004). Their use with standard forest management information has helped forest management organizations such as FUNDECOR (Fundacion para el Desarrollo de la Cordillera Volcanica Central) improve its management decision-making and enabled its operations to maintain its recognition as “sustainable” by the Forest Stewardship Council (FSC).

However, as with most forest management ecological sustainability indicators, dung beetles need to be surveyed in the field (Aguilar-Amuchastegui et al. 1999; Ghazoul 2001; Finegan et al. 2004). These surveys are limited in extent and time consuming (Lambin 1999; Farthing et al. 2001; Fahrig 2003; Foody & Cutler 2003). As the number of forest areas to be surveyed or the total area under management increases, the personnel and time required also increases and surveying becomes impractical. FUNDECOR, for example, has been struggling with this problem since 1999 as their management operations increased from 8000 to 43000 ha, making the monitoring program a logistical challenge in the face of production-related operations. FUNDECOR’s dilemma is emblematic: there is clear need for an approach to the prioritization of areas to be surveyed in the field.

Tropical forests are not static ecosystems; they manifest dynamic structural heterogeneity that results from specific natural histories of episodes of disturbance and recovery. These disturbance regimes typically create mosaics of regeneration stages (early, intermediate, advanced, and mature forest) that differ in microclimate, vegetation structure, and faunal composition, including dung beetle species community structure and composition (Morgan et al. 1994; Finegan 1996; Delgado et al. 1997; Ghazoul and Hellier 2000; Finegan & Delgado 2000; Davis et al. 2001; Weishampel et al. 2001; Andresen 2005). The horizontal distribution of regeneration stages provides a forest with the vertical and horizontal dimensions of forest structure heterogeneity (Finegan 1996; Aguilar-Amuchastegui and Henebry 2006a,b).

Forest management can change forest structure heterogeneity depending on its harvest intensity (viz., the number of trees, basal area, or cubic meters of wood removed per ha) (Delgado et al. 1997; Ghazoul and Hellier 2000; CIFOR 2000; Ghazoul 2001; Finegan et al.

2004). Accordingly, CIFOR (2000) established as one of its ecological sustainability indicators: "Changes in diversity of habitats as a result of human interventions are maintained within critical limits as defined by natural variation and/or regional conservation objectives" (I.2.1.2.). Thus, if a given management scheme maintains the relative abundance and distribution of the successional stages that provide forests with a diversity of habitats and structural heterogeneity within the limits framed by natural regimes, it may be considered sustainable. On the other hand, any scheme that fails to do so after a reasonable recovery time would be considered unsustainable.

Traditionally, forest structural heterogeneity is surveyed in the field identifying the regeneration stage of sample plots along forest inventory lines (Finegan et al. 2004). However, it can also be assessed by remote sensing (Ashton and Hall 1992; Wulder 1998; Lambin 1999; Couteron et al. 2003, 2005; Foody & Cutler 2003; Lim et al.2003; Read et al. 2003; Kalácaska et al. 2004; Lu et al. 2004; Wulder et al. 2004; Ingram et al. 2005). The optical reflectance pattern from a forested area is influenced by various structural aspects such as canopy architecture (Danson 1995), understory leaf litter (Franklin et al. 2002), biomass, age, density, mean tree height, and basal area (Lee & Nakane, 1996; Lu et al. 2004), in addition to remote sensing variables such as the sun-target-sensor geometry.

Passive optical sensors such as Landsat TM and ETM+ (Asner et al. 2002, 2004; Lu et al. 2004; Feeley et al. 2005; De Wasseige and Defourney 2004; Souza Jr. et al. 2005) and SPOT (Feeley et al. 2005; De Wasseige and Defourney 2002, 2004), active optical sensors such as LiDAR (Lefsky et al. 2002; Lim et al. 2003) and Laser Vegetation Imaging Sensor (Santos et al. 2003), and active microwave sensors such as Synthetic Aperture Radars (Israelsson and Askne 1995; Pulliainen et al. 2003) have proven useful for the characterization and monitoring of forest structure. Their use in evaluating forest stand parameters such as biomass (Wulder 1998; Lim et al.2003; Lu et al.2004), basal area, leaf area index, and average stand diameter average stand height (Lu et al. 2004), suggests that they may also be useful for evaluating CIFOR's indicator I.2.I.I. The characterization of forest canopy structure organization in space and time, by means of remote sensing, can provide of information about the state of development, function and vegetation related habitat conditions for wildlife (Lefsky et al. 2002).

There are several methods for analyzing reflectance data to assess vegetation structural parameters (Ingram et al. 2005). One of the most common approaches combines the reflectance measured at multiple spectral bands (blue, green, red, and near infra-red [NIR]) into a vegetation index (VI). Biophysical variables related to forest structure, e.g., successional and phenological stage, chlorophyll content, net primary productivity (NPP), leaf area index (LAI), and the fraction of absorbed photosynthetically active radiation (FPAR), have been shown to be related with VIs such as the Normalized Difference Vegetation Index (NDVI; Rouse et al. 1974; Baret and Guyot 1991; Veroustraete et al 1996; Fassnacht et al. 1997; Huemmrich and Goward 1997; Birky 2001; Gitelson 2004; Hoffmann 2005) and the recently developed Wide Dynamic Range Vegetation Index (WDRVI) (Gitelson 2004**;** Henebry et al. 2004;Viña et al. 2004; Viña and Gitelson 2005; Gitelson et al. 2005, 2006).

Among the methods for quantifying spatial heterogeneity, geostatistical tools such as autocorrelograms and variograms are widely used in ecology (Isaaks and Srivastava 1989; Légendre and Fortín 1989; Riera et al. 1998; He et al. 1996; Pastor et al. 1998; Goodin et al. 2004; Colombo et al. 2004). The advantage of this kind of analysis, especially variograms, lies in the fact that they provide measures of two aspects of heterogeneity: the amount of variation and the spatial extent across which that variation occurs (He et al. 1996; Henebry 1993; Goodin and Henebry 1997; Colombo et al. 2004; Goodin et al. 2004).

The present study sought to establish the relationships between (1) change in forest structural heterogeneity, (2) dung beetle community structure and composition, and (3) a key forest management variable: logging intensity. The rationale is that once the relationships between these three aspects are characterized, remote sensing of forest structure can serve as an important practical tool for guiding sustainable management of tropical forests and conserving vulnerable carbon stocks. Our main hypothesis is that managed forest areas that [do not] exhibit significant differences in forest structural heterogeneity as compared with natural areas will also [not] show significant differences in dung beetle community species diversity and composition (Figure 8.1). Using semivariography of WDRVI data from a pair of Landsat scenes, we establish the utility of remote sensing imagery in conjunction with standard forest management data to identify and priority rank areas for ecoindicator surveys. We use the differences in priority rankings and other environmental variables to model ecological dissimilarities observed between dung beetle communities present in different forest areas.

## *Methods*

The study was conducted in the Canton of Sarapiqui, in the Atlantic slope of the Cordillera Volcanica Central in Costa Rica, between 100 and 1500 meters above sea level, near Braulio Carrillo National Park (Figure 8.2). In fifteen Forest Management Units (FMU) selectively logged (≤5 trees/ha) by FUNDECOR between 1998 and 2000 and in six natural reference areas, semivariograms were used to characterize spatial heterogeneity in remotely sensed data. Data were extracted from two Landsat TM images (WRS-2 Path 15, Row 53) before logging (February 1986) and after logging (January 2001). Acquisition dates were as close to anniversary date as practical to minimize illuminations differences arising from differences in sun elevation (both angles were close to $45^o$) and to avoid bias in the spatial analysis (Goodin et al. 2004). The change in structural heterogeneity in each forest area was characterized by means of the percent change in semivariogram model parameter coefficients (the range and the ratio of the sill to the total variance; Aguilar-Amuchastegui and Henebry 2006a). This procedure enabled inclusion of the intrinsic dynamics of each forest area (Morgan et al. 1994) as well as the influence of management (Ghazoul and Hellier 2000; Ghazoul 2001; Finegan et al 2004). Each FMU was classified in terms of priority for survey based on percent changes in spatial metrics observed between periods as compared with average changes observed in natural areas. Areas within natural variation limits (within 95% confidence interval of changes observed in nearby natural areas) were classified as low priority for field survey. Areas between the 95 to 99% confidence interval considered because of recent logging (less than 5 years), were classified as moderate priority and areas beyond the 99% confidence interval were classified as high priority (Figure 8.3; cf. Aguilar-Amuchastegui and Henebry 2006a).

Due to accessibility problems and logistical constraints, only four FMU and one natural reference area inside La Selva Biological Station were surveyed for dung beetles in the field. The FMUs were selected so that they spanned a priority-for-survey gradient: Low, Moderate, and High (Figures 8.2, 8.3 and Table 8.1). All areas were less than 10 km apart and at a similar elevation, around 100 meters asl (Figure 8.2).

Ten dung-baited pitfall traps were installed every 50 meters along a 500m transect inside each forest area (Davis 2001; Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000; Finegan et al. 2004). Human feces were used as bait and ethylene glycol was used for preservation. This trap design is considered the most efficient for dung beetle collection (Silva-Milhomem et. al. 2003; Davis et al. 2001). Ethylene glycol has the advantage that it is odorless and resistant to evaporation, which can be an issue in this area where temperatures are high. Sampling in each area was carried out during three or four continuous weeks (depending on accessibility). The collected material was preliminarily sorted and identified in the field. Identifications were made by Dr. Angel Solis (Instituto Nacional de Biodiversidad [INBio] in Santo Domingo de Heredia, Costa Rica).

Species collected were classified in three main guilds based on their abundance in the natural reference area and on previous results obtained by Aguilar-Amuchastegui (1999) and Aguilar-Amuchastegui – FUNDECOR (*unpublished data*) while surveying in the same region. The C*ore* guild was constituted by species commonly recorded in these forests and from which more than five individuals were captured in the natural area. Species with fewer than five individuals captured in the natural area were allocated to the *Rare* guild. Species occurring in managed areas but not in the natural area were assigned to the *Newly Incorporated* guild.

Community structure and species richness were compared by means of observed species rank-abundance plots (Magurran 1988, 2003; Krebs 1999). Total richness and diversity estimates were obtained using the Chao estimator [8.1] (Chao 1987; Colwell & Coddington, 1994) and Fisher's alpha diversity index [8.2] (Fisher et al. 1943; Colwell & Coddington 1994; Colwell 2005), respectively:

*Chao estimator*

$$S = S_{obs} + \left( \frac{a^2}{2b} \right) \qquad [8.1]$$

where $S$ is the estimated total richness, $S_{obs}$ is the number of species observed $a$ is the number of species observed just once, and $b$ is the number of species observed just twice. (Colwell & Coddington, 1994).

*Fisher's alpha estimator*

$$\alpha = \frac{N(1-x)}{x} \qquad [8.2]$$

where $x$ as a constant estimated from the iterative solution of:

$$\frac{S}{N} = \frac{(1-x)}{x[-\ln(1-x)]} \qquad [8.3]$$

where $S$ is the observed number of species and $N$ is the number of individuals (see Magurran 1988 for more details).

Both measures were calculated using the software EstimateS v7.50 (Colwell 2005). The alpha index was chosen over other more commonly used indices, such as Shannon-Weiner or Simpson, because it is not affected by sample size (Colwell & Coddington 1994; Colwell 2005).

Community composition was compared between forest areas using the Bray-Curtis dissimilarity index [8.4] (Faith et al. 2004; Ferrier 2002; Ferrier et al. 2002):

$$BCd_{ij} = \frac{\sum_{k=1}^{S} \left| n_{ki} - n_{kj} \right|}{\sum_{k=1}^{S} \left| n_{ki} + n_{kj} \right|} \qquad [8.4]$$

where $BCd_{ij}$ is the Bray-Curtis dissimilarity between sites $i$ and $j$, $S$ is the total number of species recorded in both sites, and $n_{ki}$ and $n_{kj}$ are the number of individuals belonging to species $k$ recorded in sites $i$ and $j$, respectively (Pielou 1984).

Following Ferrier et al. (2002), we explored the possibility of modeling a linear transformation of the ecological dissimilarities matrix obtained from cross comparisons between surveyed forest dung beetles communities, using Euclidean distance matrices of environmental surrogate variables (Environmental Diversity or ED) in a linear matrix regression analysis [8.5] (Ferrier 2002; Ferrier et al. 2002; Faith 2003; Faith et al. 2004; Ferrier et al. 2004):

$$-\ln\left(1 - BCd_{ij}\right) = a + b_1\left(\left|x_{1i} - x_{1j}\right|\right) + b_2\left(\left|x_{2i} - x_{2j}\right|\right) + \ldots.. \qquad [8.5]$$

where $-ln(1\text{-}BCd_{ij})$ corresponds to the linear transformation of the compositional dissimilarity between site $i$ and site $j$, $a$ is the expected ecological dissimilarity when all the environmental explanatory variables are equal, $x_{ij}$ are the Euclidean distance values of the explanatory variables (e.g. priority for survey) between sites $i$ and $j$, and $b_i$ are the linear relation

parameter estimations obtained by means of negative log-likelihood minimization (Ferrier et al. 2002; Hilborn and Mangel 1997; Clark, 2005; Hortal and Lobo 2006).

Transformations are often needed as dissimilarities are not necessarily linear (Ferrier et al. 2002). In order to establish the degree of association of surrogate variables with ecological dissimilarities, we used a Kendall matrix correlation test (Légendre and Légendre 1998). The explanatory variables considered included five variables related to forest structure (WDRVI variance in 2001, WDRVI total variation in 2001, WDRVI range in 2001, WDRVI semivariance sill/total variation % change between 1986-2001, range % change between 1986-2001), two variables related to site survey priority (see tables 8.2 and 8.3), geographic distance between forest areas, two variables related to logging intensity (trees/ha and $m^3$/ha), and three topographic variables (mean elevation, slope, aspect) calculated from SRTM (Shuttle Radar Topography Mission) Format 3 arc-second (90 m) digital elevation model data for each forest area surveyed. Geospatial (GIS) data layers on forest area location, logging intensity, and tree locations were provided by FUNDECOR.

Because not all managed areas were logged at the same time or the same number of times (see Table 8.1), we developed a logging intensity index [8.6] to account for these two factors:

$$LI_j = le \times \left[ i_j \left( \frac{r}{y} \right) \right] \qquad [8.6]$$

where $LI_j$ is the logging intensity index in FMU $j$, $le$ is the number of logging events that occurred in the specific FMU, $i_j$ is intensity of the last logging event in trees/ha or in $m^3$/ha, $y$ is the number of years since the last logging event, and $r$ is the number of years considered for a sustainable production rotation of FMU.

The index indicates the relative logging intensity to which a forest has been exposed, based on specific logging histories. Note that as $y$ increases towards $r$, $LI$ index decreases. In the case of the FUNDECOR lowland forests, we used for $r$ an expected value of 20 years (FUNDECOR, personal communication).

Variable selection for modeling was made in order to maintain a close link between their explanatory value and the ecological and forest management rationale that supports their consideration while minimizing collinearity (Hortal and Lobo 2006). Based on this preliminary analysis, mean elevation, aspect, and geographic distance were discarded as the study areas had relatively flat relief and were relatively nearby (<10 km apart) (Hortal and Lobo 2006). Several models with different number of variables were tested in order to achieve the best possible balance between fit and number of parameters to estimate. Models using one, two and three variables were compared by means of the Akaike Information Criteria (AIC) and weight of evidence (Anderson et al. 1994; Hilborn and Mangel 1997; Burnham and Anderson 2002; Lee and Choi 2004).

### *Results*

A total of 1328 individuals from 19 different species were captured (Appendix 1). Of those, 335 individuals belonging to 11 species were captured in the natural reference area, 262 individuals from 11 species in the low priority FMU, 269 individuals from 14 species in the moderate-low priority, 197 individuals from 12 species in the moderate priority, and 265 individuals from 7 different species in the high priority for survey FMU (Table 8.4). Sampling was considered to be comprehensive as capture rates dropped dramatically after the first week (Figure 8.4) and observed richness represented over 70% of total richness estimates (Table 8.4).

Total richness and diversity estimates obtained in FMUs showed significant differences with the natural reference area with the exception of the one area classified as “low priority” for survey by Aguilar-Amuchastegui and Henebry (2006a; Figure 8.3, Tables 8.1 and 8.4). Both moderate and moderate-low priority FMUs showed significantly higher diversity and richness estimates and the high priority for survey showed significantly lower estimates (Table 8.4). The dung beetle community structure observed in managed areas was found to be more equitable than the observed in the natural area (see shallower negative slopes in FMU’s species rank-abundance plots in Figure 8.5; Magurran 1988; Krebs 1989).

Community composition dissimilarities between natural and managed areas as measured with the Bray-Curtis dissimilarity index showed an apparent increasing trend as priority for survey increased (see first column of Table 8.5). Table 8.6 shows the strength of the correlation between transformed Bray-Curtis dissimilarities of dung beetle communities and various environmental surrogates. Strong, significant correlations were found with both measures of forest structural heterogeneity (WDRVI total variation in 2001: $\rho=0.471$; range 2001: $\rho=0.535$), with both transformations of the priority-for-survey ranking (linear transform: $\rho=0.675$; binary transform: $\rho=0.546$), and with the logging intensity index measured in trees per hectare ($\rho=0.349$).

Matrix linear regression modeling revealed that the single variable with the highest explanatory power (as measured by AIC and weight of evidence) was the linear transformation of survey priority, followed by the binary transformation, logging intensity index in trees per hectare, and mean slope. The best overall model (delta AIC=0 and weight of evidence =0.3) included all three variables but was closely followed by models including only the first and either of the second or third variable (cf. weights of evidence in Table 8.7). The root mean square error of the best model was 0.078 or about 8% of observed average (Figure 8.6).

## *Discussion*

### *Forest logging intensity threshold*

Previous studies have found that when logging intensity in tropical forests is around four trees per hectare, no significant structural differences are observed following harvest (Pinard and Cropper 2000; Pinard et al. 2000; Pinard and Putz 1996). Our previous results (Aguilar-Amuchastegui and Henebry 2006a, b) and the modeling presented here support the existence of this threshold. The only areas that showed significant differences in forest spatio-temporal heterogeneity were areas with logging intensities greater than four trees per hectare (Aguilar-Amuchastegui and Henebry 2006a). When considering dung beetle community structure and composition, previous studies have found that logging intensities less than four trees per hectare generate no reduction in tropical rainforest dung beetles species richness and diversity (Davis 2001; Davis et al. 2001; Scheffler 2005). That is also our case: the high priority-for-survey FMU was subjected to a logging intensity of five trees per hectare and exhibited significant differences in forest structural heterogeneity and dung beetle total species richness and diversity. The remaining FMUs surveyed, logged at lower intensities, showed neither significant losses in dung beetle species richness and diversity nor significant differences in forest structural heterogeneity.

These results were not unexpected, as it has been shown that low impact forest logging as opposed to reducing total dung beetles species richness and diversity makes dung beetle communities more equitable (Figure 8.5; Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000; Aguilar-Amuchastegui and FUNDECOR *unpublished data*; Davis et al. 2001; Scheffler 2005). This pattern results from a rearrangement of community structure caused by the reduction in the dominance of natural areas "*Core*" species (species commonly captured in the interior of this type of forests) (Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000; Aguilar-Amuchastegui-FUNDECOR *unpublished data*), enabling "*Newly Incorporated*" species to occur (Nummelin and Hanski 1989; Nummelin 1998; Aguilar-Amuchastegui 1999; Aguilar-

Amuchastegui et al. 2000; Figure 8.7). Changes in forest structural heterogeneity cause an increased overlap of species which ranges are usually separated (Davis et al. 2001; Scheffler 2005). Logging related forest structural disturbances such as logging trails allow the entrance of species preferring edges and open areas into the dung beetle community with the FMU (Aguilar-Amuchastegui et al. 2000; Davis et al. 2001; Scheffler 2005). Such is the case of *Canthidium haroldi* Preudhomme 1886, a forest interior species linked with white-faced monkey (*Cebus capucinus* L.) and the dominant dung beetle species in natural areas in the region (Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000; Solis and Kohlman 2004; Wehncke and Dalling 2005). While *C. haroldi* showed a reduction in dominance as priority for survey increased, *Canthon aequinoctialis* Harold 1868, a generalist species typically recorded in forest edge environments, showed the opposite trend (Table 8.8 and Appendix 2; Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000; Solis and Kohlmann 2002). Based on these results and previous work on logging impacts on forest structure and dung beetle community structure and composition, a threshold of four trees per hectare may be established between sustainable and unsustainable logging intensities for Neotropical lowland rainforests.

### *Forest structural heterogeneity as means for establishing priority for field surveys of ecoindicators of sustainability*

Previous studies have established how dung beetle community structure and composition are related to forest structure and how forest management intensity can affect them (Halffter and Favila 1993; Favila and Halffter 1997; Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000; Davis et al. 2001; Pineda et al. 2005; Scheffler 2005; Andresen 2005; Vernes et al. 2005). In our results, positive significant correlations found between dung beetle community composition dissimilarities and forest structural heterogeneity (as measured by WDRVI total variation and range in 2001), on the one hand, and a logging intensity index, on the other, provide evidence of interactions among these three aspects of disturbance and response: logging regime, forest structural heterogeneity, and dung beetle community structure and composition (Table 8.6).

The significant correlation found between the priority-for-survey ranking variable, derived from changes in forest structural heterogeneity between 1986 and 2001 within each FMU compared to natural reference areas, and dung beetle community dissimilarities (Table 8.6) points to the dynamic aspect of the interactions. The priority-for-survey ranking variable showed the highest explanatory power between the variables tested when modeling the compositional dissimilarities observed between the dung beetle communities present in each forest area (Table 8.7). The disturbance regime, represented by the logging intensity index in trees per hectare, plays also a significant role in explaining compositional dissimilarities (Table 8.7). Both the disturbance history and change in forest structural heterogeneity are included in those models that showed the better fits and weights of evidence for explaining the observed dissimilarities in dung beetle communities (Table 8.7, Figure 8.6).

### *Environmental diversity as means of modeling ecological dissimilarities between FMU and natural reference areas*

Modeling of ecological diversity based on environmental surrogate variables (environmental diversity) has been proposed for use in the identification of priority areas for field survey of biodiversity for inventory purposes and/or for the design of protected areas (Araújo et al 2001, 2003; Ferrier et al 2002, Ferrier 2002; Faith 2003; Faith et al. 2004; Ferrier et al. 2004; Hortal and Lobo 2005). While this approach has already been used successfully in Australia (Ferrier 2002; Faith 2003), other studies have shown that it does not always provide better results than the ones obtained by a random selection of sampling sites (Araújo et al 2001; Araújo et al 2003). Referring to Araújo et al. (2001, 2003), Hortal and Lobo (2005) point out that such lack of explanatory power may be related to what they call “contingent factors” that include historical events that can modify the spatial pattern of biodiversity and that are “difficult to express”. The explanatory power of “current” environmental diversity is predicated on the historical relationships (Holling 1992). Places with similar present conditions may hold different communities due to specific histories of land use and/or disturbance. Such contingency underlines the need for a dynamic baseline (Hortal and Lobo 2005). Our approach allows this problem to be addressed by modeling ecological diversity using some of those contingency factors that cause current environmental diversity. Forest disturbance history and changes in forest structural heterogeneity, represented by the logging intensity index and priority-for-survey ranking, respectively, provide a dynamic baseline for environmental diversity and so include “contingent” site history factors. Using these two variables we were able to model dung beetle ecological dissimilarities as observed between FMUs and natural reference areas (Table 8.7 and Figure 8.6).

Our findings in this pilot study demonstrate how the ecological interactions between dung beetles and forest structural heterogeneity and disturbance regime enable FMUs to be assessed remotely, evaluated, and prioritized for field surveys. Furthermore, they show how readily available geospatial and remote sensing data can be used to predict ecological dissimilarities between areas with different forest management. However, it is clear that further work is needed in additional regions, with different forest management regimes, in different forest types, and with additional ecoindicators to test, evaluate, and tune the methodology.

### *Conclusions*

The present work establishes links existing between dung beetle community structure and composition, forest structural heterogeneity and its dynamic change, and one of the main forest management variables: logging intensity. Our central hypothesis was supported: managed forest areas that [did not] exhibit significant differences in forest structural heterogeneity as compared with natural areas also [did not] show significant differences in dung beetle community species diversity and composition. More specifically, we found that areas ranked as low priority for survey based on image analysis showed no significant difference in dung beetle species richness or diversity from natural references. Further, we found that in areas ranked as moderate or moderate-low priority for survey, there were significant increases in dung beetle species richness and diversity over the low and reference areas. Finally, the dung beetle community composition in the high priority for survey category was significantly less rich and diverse than any of the other categories.

A logging intensity threshold has been identified. Even though each forest management unit is unique and similar logging intensities may have differential impacts, basic evidence of a logging intensity threshold around four trees per hectare has been provided with two different CIFOR indicators: “The change in diversity of habitats as a result of human interventions is maintained within critical limits as defined by natural variation and/or regional conservation objectives” (I.2.I.2 ) and “The richness/diversity of selected groups show no significant change” (I.2.I.4).

A prioritization system based on remote sensing that we developed earlier (Aguilar-Amuchastegui and Henebry 2006a) showed significant agreement with dung beetle community structure and composition, a well-known ecoindicator. We provide basic evidence for considering that approach as a promising means for forest managers and conservation agencies to target priority areas for ecological field surveys and it shows promise as an important practical tool for guiding sustainable management of tropical forests and conserving vulnerable carbon stocks. However, further work with other ecoindicator groups such as mammals, birds, and butterflies and in other types of tropical forests will be required before this approach can be used for decisions about the sustainability of management practices.

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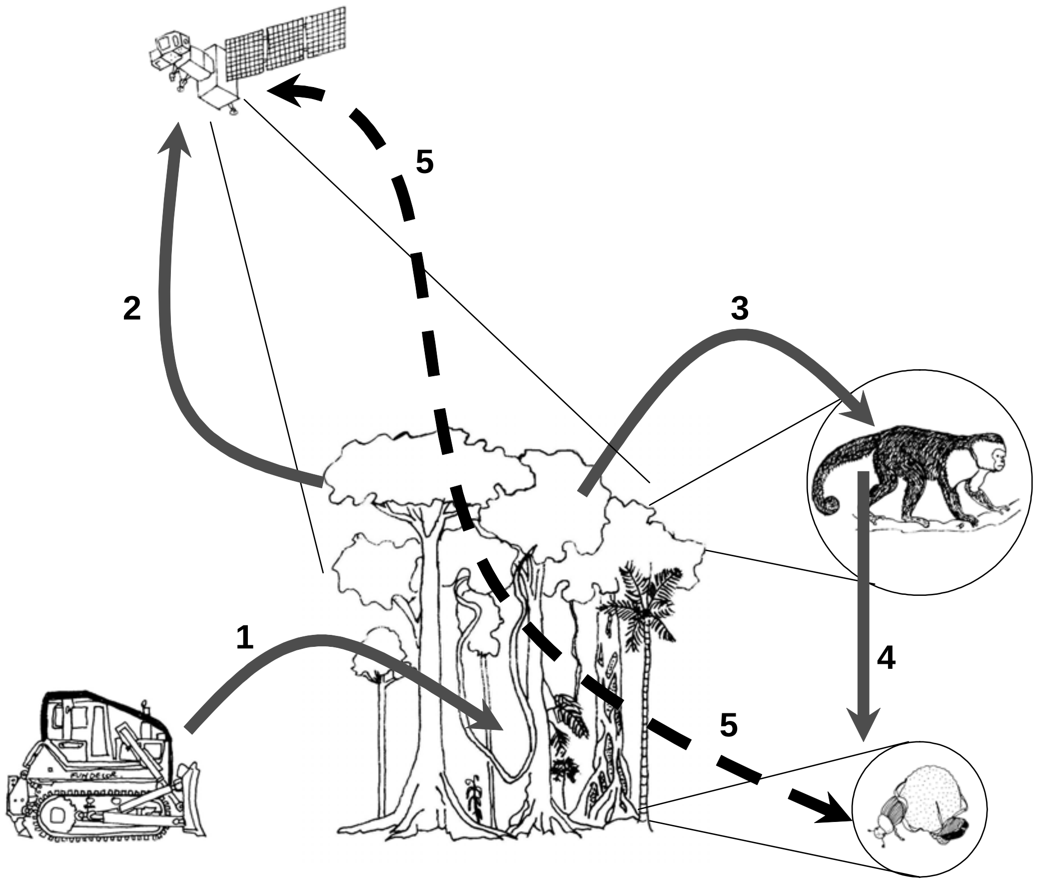

**Figure 8.1.** Illustrated hypothetical linkages from dung beetle to remote sensor: (**1**) Forest management can change forest structural heterogeneity; (**2**) Forest structural heterogeneity and its change can be measured by remote sensing using semivariography of vegetation indices; (**3**) Forest structural heterogeneity influences habitat quantity and quality for forest vertebrate fauna; (**4**) Forest vertebrate fauna composition influences dung production and thus dung beetle community structure and composition; and (**5**) the hypothesis tested in this paper: *Managed forest areas that [do not] exhibit significant differences in forest structural heterogeneity as compared with natural areas will also [not] show significant differences in dung beetle community species diversity and composition*.

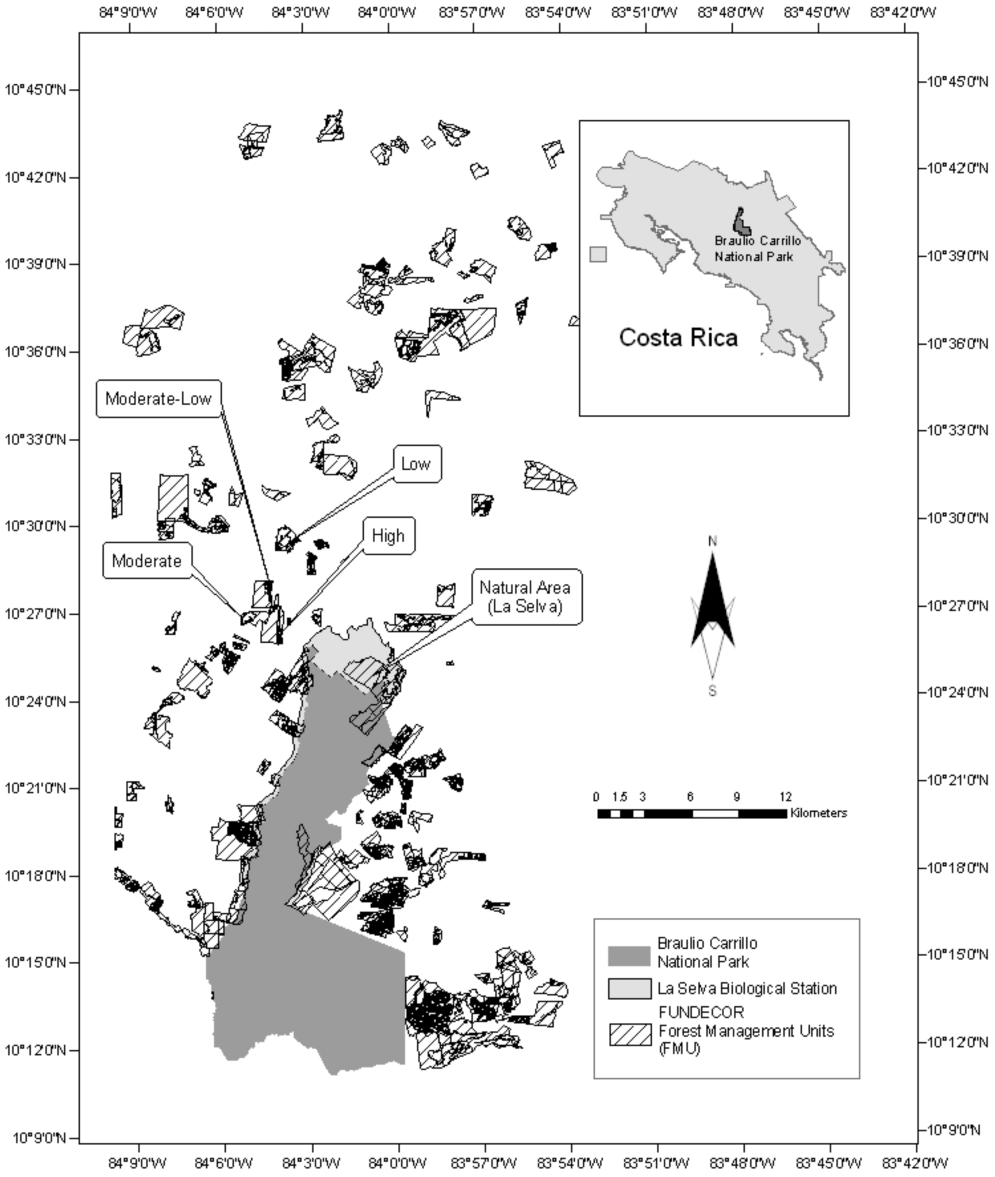

**Figure 8.2.** Mapped locations of the surveyed forest areas and their regional context. The natural area is located inside La Selva Biological Station from the Organization for Tropical Studies (OTS).

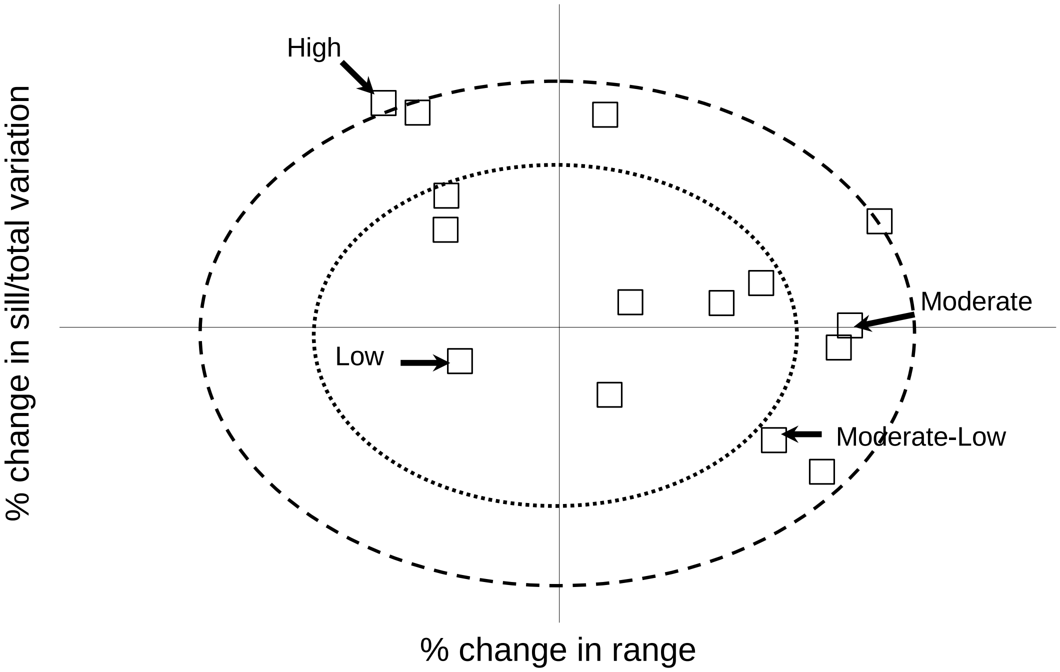

**Figure 8.3.** Change in forest structural heterogeneity between 1986 and 2001 as described by changes in the range and in the ratio of sill to total variation from semivariogram analyses of Wide Dynamic Range Vegetation Index values from fifteen FUNDECOR forest management units. The axis origin and elliptical boundaries show the average changes observed in the six natural forests with their 95% and 99% confidence intervals. The location of a specific forest management unit with respect to these thresholds enables an interpretation in terms of priority for survey in the field. The FMUs indicated by arrows were selected for field surveying of dung beetles community structure and composition (modified from Aguilar-Amuchastegui and Henebry – 2006a with permission from IEEE).

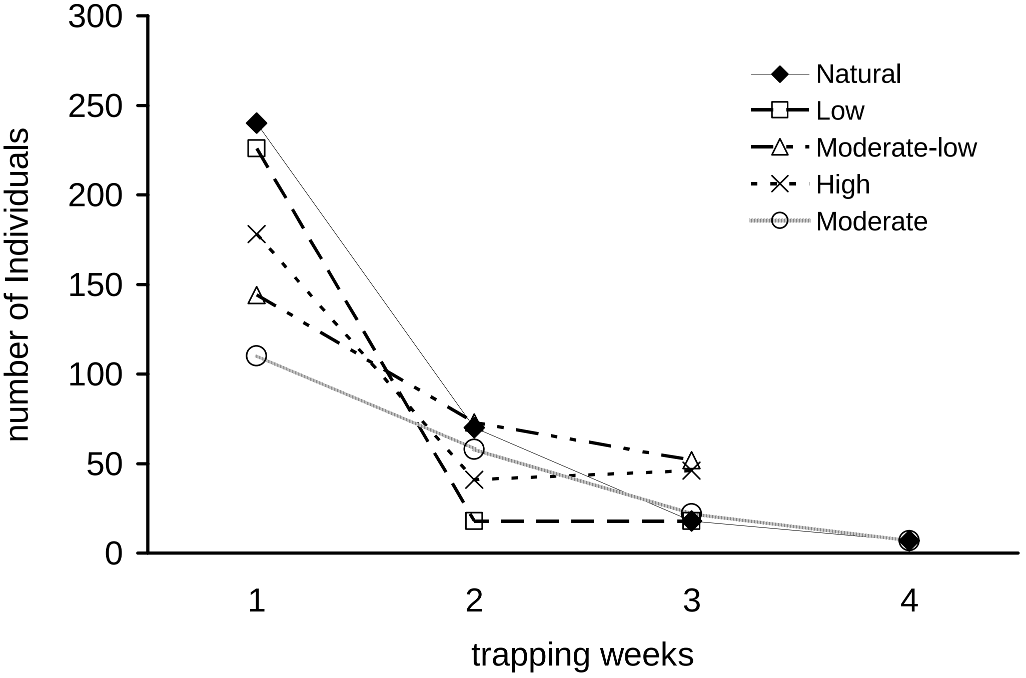

**Figure 8.4.** Number of individuals captured per week in each of the surveyed forests.

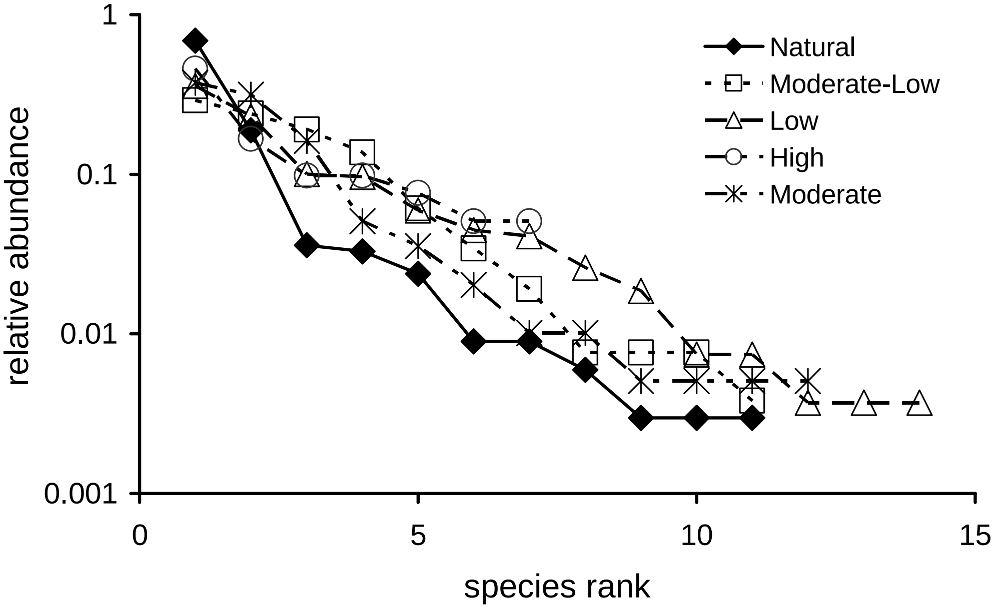

**Figure 8.5.** Species rank-abundance plots as observed in each forest area (note log scale of ordinate). A shallower negative slope indicates a more even, equitable community composition because dominance by the most abundant species decreases (Magurran 1988, Krebs 1999). A longer curve implies a higher richness. An area with higher diversity levels will show both longer and shallower distributions. Such is the case of both moderate and moderate-low survey priority sites compared to the natural reference area.

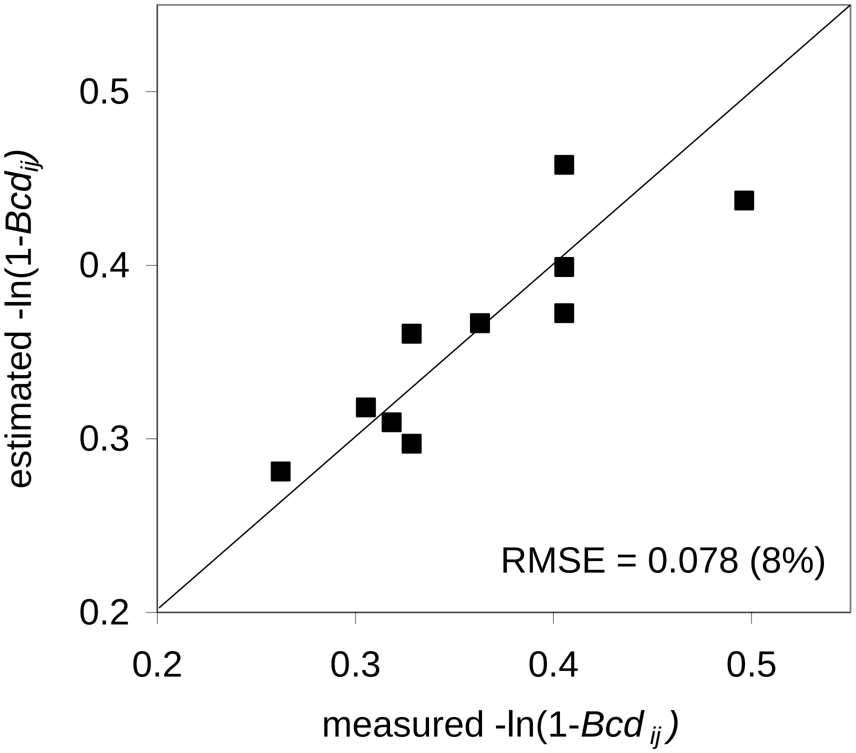

**Figure 8.6.** Observed and estimated linear transformations of Bray-Curtis dissimilarities between dung beetle communities. Estimates were made using distance linear matrix regression modeling following Ferrier et al. (2002). The variables used include the linear transformation of priority for survey, logging intensity index in trees/ha, and mean slope (see Table 8.7).

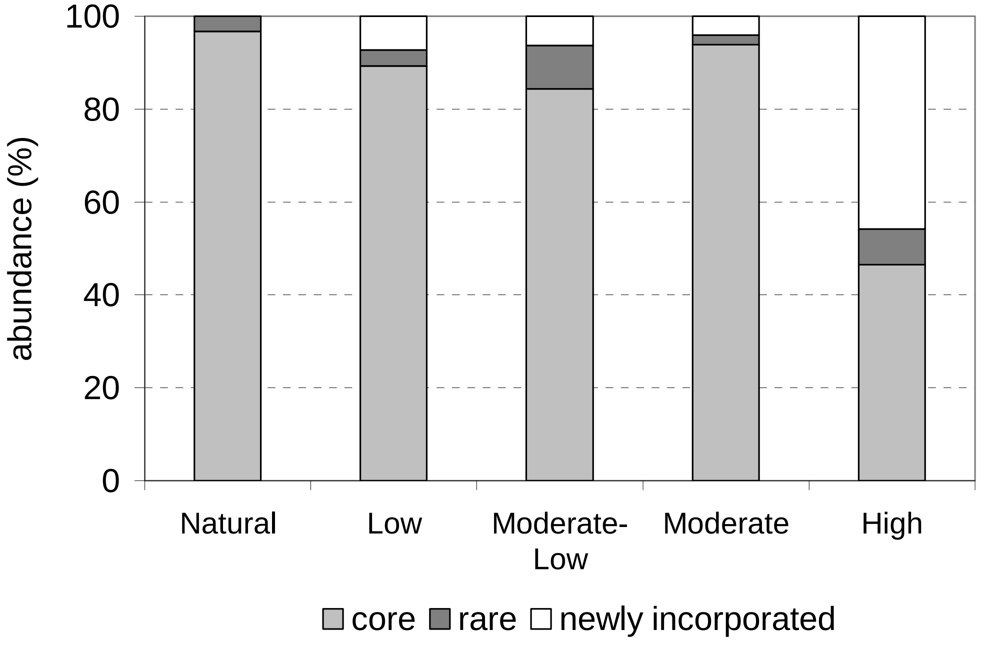

**Figure 8.7.** Dung beetle guild community structure shift as a result from logging disturbance. Natural areas dung beetles "*Core Fauna*" (species commonly recorded in these forests [Aguilar-Amuchastegui 1999; Aguilar-Amuchastegui et al. 2000; Solis 2006 *personal communication*] from which more than 5 individuals were captured in the natural reference area) dominance decreases with logging disturbance, enabling "*Newly Incorporated*" species not present in the natural area to occur. *"Rare Species"* represent less than 10% of total individuals captured. Major changes occur in the high priority for survey area. Changes include differences in dominant species, the incorporation of new species and the loss of other species; depending on logging intensity (see Appendix 2).

**Table 8.1.** Forest Management Unit (FMU) priority for field survey level and logging intensity.

| FMU priority | Area (ha) | Times logged | Last logging event | | | Logging Intensity Index | |
|---|---|---|---|---|---|---|---|
| | | | year | intensity | | | |
| | | | | trees/ha | $m^3$/ha | trees/ha | $m^3$/ha |
| Low | 45 | 1 | 1998 | 1.8 | 4 | 10.3 | 23 |
| Moderate-Low | 43 | 1 | 2000 | 1.3 | 17 | 10.4 | 136 |
| Moderate | 40 | 2** | 2000 | 3 | 6 | 40 | 50 |
| High* | 22 | 1 | 1998 | 5 | 15 | 34.3 | 178 |

* Area showing significant differences with respect to natural variation limits (outside 99%CI).

** Logged before it came under FUNDECOR management in 1986.

**Table 8.2.** Priority for survey linear transformation distance matrix. As priority between areas increase, distance increases in a linear manner.

| Forest | Natural | Low | Moderate-Low | Moderate |
|---|---|---|---|---|
| Low | 0.0 | | | |
| Moderate-Low | 1.0 | 1.0 | | |
| Moderate | 1.5 | 1.5 | 0.5 | |
| High | 2.0 | 2.0 | 1.0 | 0.5 |

**Table 8.3.** Priority for survey binary transformation distance matrix. Areas under the same priority for survey are given a distance of 0. Areas in different priority levels are given a distance of 1.

| Forest | Natural | Low | Moderate-Low | Moderate |
|---|---|---|---|---|
| Low | 0 | | | |
| Moderate-Low | 1 | 1 | | |
| Moderate | 1 | 1 | 0 | |
| High | 1 | 1 | 1 | 1 |

**Table 8.4.** Number of individuals captured, observed species, and total species richness and diversity estimates for each of the forest areas surveyed.

| Forest | Individuals | Species richness | | Diversity estimates |
|---|---|---|---|---|
| | | Observed | Estimated total +/- 2SE (Chao mean) | Fisher's alpha +/- 2SE (mean value) |
| Natural | 335 | 11 (94%)* | 11.73 a | 2.19 a |
| Low | 262 | 11 (97%)* | 11.32 a | 2.32 a |
| Moderate-Low | 269 | 14 (94%)* | 14.97 b | 3.14 b |
| Moderate | 197 | 12 (71%)* | 16.83 b | 2.85 b |
| High | 265 | 7 (100)* | 7 c | 1.32 c |

* In parentheses is the percentage of estimated total richness represented by actual observed richness.

**Table 8.5.** Bray-Curtis dissimilarity distances ($BCd_{ij}$) x 100 as observed between surveyed forests based on dung beetles community species composition.

| Forest | Natural | Low | Moderate-Low | Moderate |
|---|---|---|---|---|
| Low | 27.3 | | | |
| Moderate-Low | 28.0 | 28.0 | | |
| Moderate | 30.4 | 39.1 | 23.0 | |
| High | 33.3 | 33.3 | 33.3 | 26.3 |

**Table 8.6**. Kendall matrix correlation coefficients between the Euclidean distances of environmental surrogate variables and the composition of dung beetle communities based transformed Bray-Curtis dissimilarities.

| Environmental surrogate variables | Kendall matrix correlation coefficient |
|---|---|
| Geographic distance | 0.116 |
| WDRVI variance 2001 | 0.163 |
| WDRVI total variation 2001 | * 0.471 |
| WDRVI range 2001 | ** 0.535 |
| WDRVI semivariance sill/total variation change (%) 2001-1986 | -0.116 |
| WDRVI semivariance range change (%) 2001-1986 | -0.023 |
| Elevation mean | 0.210 |
| Elevation variance | 0.023 |
| Slope mean | 0.210 |
| Slope variance | -0.023 |
| Aspect mean | -0.070 |
| Logging intensity index ($m^3$/ha) | 0.023 |
| Logging intensity index (trees/ha) | * 0.349 |
| Priority for survey (linear) | ** 0.675 |
| Priority for survey (binary) | * 0.546 |

* Significant at 90%; ** Significant at 95%.

**Table 8.7.** Modeling of dung beetle community Bray-Curtis dissimilarity linear transformations: variables used, negative log-likelihood and Akaike Information Criteria (AIC) for model selection. A lower AIC value indicates a better model. Delta AIC =0 indicates the model with the best balance between explanatory power and parameter coefficient estimation error.

| | | 1 | | | | | 2 | | | 3 |
|---|---|---|---|---|---|---|---|---|---|---|
| Variables used | | Priority (linear) | Priority (binary) | Logging ($m^3$/ha) | Logging (trees/ha) | Mean slope | Priority (linear) | Priority (linear) | Priority (linear) | Priority (linear) |
| | | | | | | | Logging ($m^3$/ha) | Logging (trees/ha) | Mean slope | Logging (trees/ha) |
| | | | | | | | | | | Mean Slope |
| Parameter values | a | 0.284 | 0.289 | 0.800 | 0.314 | 0.328 | 0.320 | 0.234 | 0.282 | 0.212 |
| | $b_1$ | 0.071 | 0.086 | $1*10^{-4}$ | 0.002 | 0.012 | $7*10^{-4}$ | 0.079 | 0.068 | 0.068 |
| | $b_2$ | | | | | | 0.097 | 0.016 | $1*10^{-4}$ | 0.001 |
| | $b_3$ | | | | | | | | | 0.019 |
| Neg Log-likelihood | | 10.974 | 12.233 | 14.545 | 13.497 | 13.792 | 10.971 | 7.680 | 7.882 | 6.715 |
| AIC | | 25.948 | 28.467 | 33.09 | 30.9945 | 31.585 | 27.943 | 21.430 | 21.764 | 21.359 |
| Delta AIC | | 4.588 | 7.107 | 11.731 | 9.635 | 10.225 | 6.583 | 0.070 | 0.405 | 0.0 |
| Weight of Evidence | | 0.03 | 0.008 | 0.0008 | 0.002 | 0.0018 | 0.011 | 0.29 | 0.25 | 0.30 |

**Table 8.8.** Percentage of total individuals collected in each forest areas that belonged to *Canthidium haroldi* and *Canthon aequinoctialis*.

| | Forest | | | | |
|---|---|---|---|---|---|
| Species | Natural | Low | Moderate-Low | Moderate | High |
| *Canthidium haroldi* | 69.2 | 29.0 | 23.0 | 3.7 | 1.5 |
| *Canthon aequinoctialis* | 0 | 0 | 0.37 | 2.1 | 47.5 |

# Chapter 9: General conclusions

In the present dissertation we sought to establish whether remotely-sensed imagery can be used in conjunction with standard forest management information to identify priority areas for field surveys of well known ecological sustainability indicators such as dung beetle species diversity and composition. We sought to determine relationships between observable heterogeneities in forest canopy reflectance, forest management practices, and dung beetle community structure and composition. The long-range goal: to contribute to the development, test, and dissemination of a methodology that enables tropical forest managers to use the relationship between the spatial heterogeneity of canopy reflectance and dung beetle community diversity and composition as a means of programming field surveys of biodiversity indicators in general. In order to achieve this goal, specific steps were taken to test the hypothesis that needed to be tested with respesct to (1) quantification of the spatial heterogeneity of forest structure, and (2), the relationshipsexisting between forest structure heterogeneity and dung beetle community structure and composition. Here I present the overall general conclusions resulting from the results presented in chapters 6 to 8:

1. *Quantification of the spatial heterogeneity of forest structure.*

Research hypothesis:

*More [less] intensely managed forest stands will exhibit [no] significant differences in the spatial heterogeneity of canopy reflectance as observed by spaceborne sensors in comparison with undisturbed control stands.*

Specific conclusions:

This research demonstrated that forest structural heterogeneity can be measured by semivariogram analysis of remote sensing-derived vegetation indices. The WDRVI proved to be useful for the quantification of tropical rainforest structure heterogeneity thanks to the dynamic range enhancement provided by the $\alpha$ factor, which can be optimized specifically for each forest area and image acquisition.

Comparisons between the dynamic changes of semivariance dimensions of the WDRVI as observed in natural and managed areas allowed identification of managed areas that behaved out of what would be considered natural variation limits and classification of priorities for field surveys of ecological sustainability indicators. These areas were the most intensely logged, with a logging intensity above 4 trees cut per hectare. More intensely logged areas exhibited significant differences in the spatial heterogeneity of forest structure heterogeneity which allowed accetinging this hypothesis.

2. *Relationships between forest structure heterogeneity and dung beetle community structure and composition and model dung beetle diversity as a function of canopy reflectance heterogeneity.*

Research hypothesis:

*Managed forest stands that do [not] exhibit significant differences in the spatial heterogeneity of canopy reflectance will also show [no] significant differences in dung beetle species diversity and community composition.*

Specific conclusions:

Dung beetle community structure and composition can be affected by forest logging intensity and derived changes in forest structure heterogeneity. Forest areas showing significant differences in forest structure heterogeneity dynamic change as compared with natural areas showed significant differences in dung beetle community structure and composition. Forest structure heterogeneity dynamic change as characterized by means of semivariogram analysis of the WDRVI, and forest logging intensity proved to be good predictors of ecological dissimilarities as observed between dung beetle communities present in different forest management units. These results provided evidence to accept the second hypothesis.

Following the proposed rationale, relationships between canopy spatial heterogeneity and dung beetle biodiversity were characterized. Remote sensing of forest structure was shown to be useful as a practical tool for guiding sustainable management of tropical forests and conserving vulnerable carbon stocks. This provided of evidence that gave support to the central hypothesis**:** the spatial heterogeneity of tropical rainforests is related to the spatial heterogeneity of forest successional stages, and to vertebrate fauna diversity and composition, which is, in turn, related to dung beetle community structure and composition due to well-documented specializations of dung beetle species with the particular dung producers (Figure 1).

Finally, it is important to emphasize that this approach does not pretend and cannot replace field surveys of biodiversity indicators. Its only purpose is to provide managers with a tool that they may ba able to use whenever, due to logistical constraints, they are in need of establishing priorities when programming field surveys of indicators of ecological sustainability. A tool that allows them the an approximation towards the optimal use of their resources in order to generate good quality information for decision making purposes.

# Chapter 10: Recommendations

As mentioned in the introductory chapter, the long-range goal of this dissertation was to develop, test, and disseminate a methodology that may enable tropical forest managers and forest managers in general, to use the relationship between forest structure heterogeneity and the biodiversity present within it as a means of scheduling field surveys of biodiversity indicators in general. The results obtained have characterized the relationship with only one component of forest biodiversity; dung beetle community structure and composition. Even though dung beetle community structure and composition has been proved to be an excellent indicator of tropical forest biodiversity, this is still a pilot study that needs to be complemented with additional work with other indicator groups, in other forests. Only then, if the results are similar, will enough evidence exist to provide the basis for the development of a practical way of using information derived from spaceborne sensors for forests management sustainability assessment. Such a tool that forest managers, conservation agencies and governmental agencies will be able of using it decision making, policy making and impact monitoring. In order to achieve such a goal the recommendations are:

1. Perform surveys of additional indicators in the areas surveyed in the present work. This will provide a more comprehensive picture of the biodiversity present in each forest and on the relation existing between CIFOR's indicator I.2.1.2. and other indicators of ecological sustainability (see Appendix 1). FUNDECOR has an extensive experience surveying vegetation communities, butterflies and birds. All these groups are considered good indicators as Dung Beetles. Such experience should allow an easy evaluation and complementation of the results obtained here.
2. Similar evaluations need to be made in additional areas. These can include those FUNDECOR FMU that could not be surveyed in the field during the present research making a larger sample number and/or additional areas from different forest management operations in other regions in the tropics and/or different types of forests. Large forest management concessions as the ones existing in Bolivia and Brazil should be of great interest.

**3.** Even though the WDRVI proved to be very useful for tropical forest structural heterogeneity monitoring thanks to the signal amplifying effect that the $\alpha$ coefficient has on the index, testing additional vegetation indices might prove wise.

**4.** Data from sensors other than Landsat TM need to be tested. The problems shown by an aging TM sensor and a dysfunctional ETM+ make it necessary to consider in the use of other alternatives such as EO-1 ALI, SPOT and even MODIS for large forested areas.

**5.** Imagery analysis should be made periodically. The periodicity should be established based on the duration of the cut cycle assigned for each specific FMU. As a rule of thumb surveys should be made every 5 years in order to follow forest recovery after logging. As cloud cover is always an issue and imagery acquisition is constant, managers should always at the lookout for new images that are cloud free and therefore usable for forest heterogeneity dynamic change survey.

**6.** The logging intensity threshold here identified needs to be considered with care. The results obtained come from a very small sample size. Additionally, as mentioned in the document, each forest management unit, because of its unique characteristics will respond differently to different logging intensities. The current research was made in areas in which logging is mechanized, which implies additional disturbance apart from the one directly caused by tree logging. Non mechanized operations will certainly show a different logging intensity threshold.

**Appendix 1.** CIFOR criteria and indicators generic template for ecological sustainability (P=Principle; C= criteria; I=Indicator; V=verifier) (taken from CIFOR. [2000] Criteria and Indicators for the Sustainable Forest Management: Generic Template. URL:<http://www.cifor.cgiar.org/acm/methods/toolbox2.html>).

| P.2. | Maintenance of Ecosystem Integrity |
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| C.2.1 | The processes that maintain biodiversity in managed forests (FMUs) are conserved. |
| I.2.1.1 | Landscape pattern is maintained |
| V.2.1.1.1 | FMU compiles information on areal extent of each vegetation type in the intervention area compared to area of the vegetation type in the total FMU |
| V.2.1.1.2 | Number of patches of each vegetation type at the FMU is maintained within natural variation |
| V.2.1.1.3 | Largest patch size of each vegetation type is maintained within critical limits |
| V.2.1.1.4 | Area weighted patch size is maintained within critical limits |
| V.2.1.1.5 | Contagion index of the degree to which vegetation types are aggregated, is maintained within |
| V.2.1.1.6 | Dominance of patch structure does not show significant change as compared to unlogged site |
| V.2.1.1.7 | Fractal dimension of patch shape is maintained within critical limits |
| V.2.1.1.8 | Average, minimum, and maximum distance between two patches of the same cover type are maintained within natural variation4 |
| V.2.1.1.9 | Percolation index, specifying landscape ‘connectedness’, is maintained within critical limits |
| V.2.1.1.10 | Linear measures of the total amount of edge of each vegetation type exist |
| V.2.1.1.11 | Amount of edge around the largest patch does not show significant change as compared to undisturbed forest |
| ***I.2.1.2*** | ***Change in diversity of habitat as a result of human interventions are maintained within critical limits as defined by natural variation and/or regional conservation objectives*** |
| *V.2.1.2.1* | *Vertical structure of the forest is maintained within natural variation* |
| *V.2.1.2.2* | *Size class distribution does not show significant change over natural variation* |

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| *V.2.1.2.3* | *Frequency distributions of leaf size and shape are maintained within natural variation* |
| *V.2.1.2.4* | *Frequency distribution of phases of the forest regeneration cycle is maintained within critical limits* |
| *V.2.1.2.5* | *Canopy openness in the forest understorey is minimized* |
| *V.2.1.2.6* | *Other structural elements do not show significant change* |
| *V.2.1.2.7* | *The distribution of above ground biomass does not show significant change as compared to undisturbed forest* |
| ***I.2.1.3*** | ***Community guild structures do not show significant changes in the representation of especially sensitive guilds, pollinator and disperser guilds*** |
| V.2.1.3.1 | Relative abundance of seedling, saplings and poles of canopy tree species belonging to different regeneration guilds does not show significant change as compared to undisturbed forest |
| V.2.1.3.2 | The abundance of selected avian guilds is maintained within natural variation |
| V.2.1.3.3 | The abundance of nests of social bees is maintained within natural variation |
| V.2.1.3.4 | The abundance of seed in key plant species does not show significant change as compared to undisturbed forest |
| V.2.1.3.5 | Fruiting intensity in known bat-pollinated tree species does not show significant change as compared to undisturbed forest |
| V.2.1.3.6 | The abundance and activity of terrestrial frugivorous mammals is maintained within critical limits |
| *V.2.1.3.7* | *The diversity of forest floor invertebrate communities does not vary significantly between logged and undisturbed forest* |
| ***I.2.1.4*** | ***The richness/diversity of selected groups show no significant change*** |
| *V.2.1.4.1* | *Species richness of prominent groups is maintained or enhanced* |
| V.2.1.4.2 | Number of different birdcalls do not vary significantly as compared to unlogged site |
| V.2.1.4.3 | Number of large butterfly species is maintained within natural variation |
| V.2.1.4.4 | Numbers of species removed from the forest for sale in local markets |
| V.2.1.4.5 | Lists of selected groups of species, compiled by acknowledged experts, do not show significant change |

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| *V.2.1.4.6* | *Temporal change in species richness is not significant* |
| V.2.1.4.7 | Time series of the ratio of composition of mature forest species to secondary growth species shows no significant change |
| *V.2.1.4.8* | *The spatial diversity of selected groups is maintained within natural variation* |
| I.2.1.5 | Population sizes and demographic structures of selected species do not show significant change, and demographically and ecologically critical life-cycle stages continue to be represented |
| V.2.1.5.1 | The absolute population size of selected species is maintained within natural variation |
| V.2.1.5.2 | Temporal change in the population size is not significant |
| V.2.1.5.3 | Tree age or structure does not show significant change as compared to undisturbed forest |
| V.2.1.5.4 | Population growth rate does not show significant change as compared to undisturbed forest6 |
| V.2.1.5.5 | Spatial structure of population is maintained within natural variation |
| I.2.1.6 | The status of decomposition and nutrient cycling shows no significant change |
| V.2.1.6.1 | Standing and fallen dead wood does not show significant change as compared to undisturbed forest |
| V.2.1.6.2 | State ofdecay of all dead wood does not show significant change as compared to undisturbed forest |
| V.2.1.6.3 | Abundance of small woody debris does not show significant change as compared to undisturbed forest |
| V.2.1.6.4 | Depth of litter/gradient of decomposition does not vary significantly between undisturbed and logged sites |
| V.2.1.6.5 | Abundance of decomposer organisms is maintained within natural variation |
| V.2.1.6.6 | Decomposition rate on the forest floor does not show significant change |
| V.2.1.6.7 | Soil conductivity and pH do not show significant change as compared to unlogged site |
| V.2.1.6.8 | Soil nutrient levels are maintained within critical limits |
| I.2.1.7 | There is no significant change in the quality and quantity of water from the catchment |

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| V.2.1.7.1 | Abundance and diversity of aquatic organisms is maintained within critical limits |
| V.2.1.7.2 | Chemical composition of stream water does not show significant variation as compared to unlogged forest |
| V.2.1.7.3 | Decomposition rate of the stream water does not show significant change as compared to unlogged forest |
| V.2.1.7.4 | Stream flow does not show significant change as compared to the flow in the unlogged site |
| C.2.2 | No chemical contamination to food chains and ecosystem |
| I.2.2.1 | Ecosystem function is maintained |
| I.2.2.2 | Ecologically sensitive areas, especially buffer zones along watercourses, are protected |
| I.2.2.3 | Representative areas, especially sites of ecological importance, are protected and appropriately managed |
| I.2.2.4 | Rare or endangered species are protected |
| I.2.2.5 | Erosion and other forms of soil degradation are minimised |
| C.2.3 | Conservation of the processes that maintain genetic variation7 |
| I.2.3.1 | Levels of genetic diversity are maintained within critical limitsDemographic verifiers: |
| V.2.3.1.1 | Census number of sexually mature individuals is above critical absolute values |
| V.2.3.1.2 | Census number of reproducing individuals is above critical absolute values |
| V.2.3.1.3 | Coefficient of phenotypic variation is higher or not significantly different from reference population |
| Genetic verifiers: | |
| V.2.3.1.4 | Number of alleles is maintained |
| V.2.3.1.5 | Gene diversity is maintained |
| V.2.3.1.6 | Genetic variation is maintained |
| I.2.3.2 | There is no directional change in genotypic frequencies |
| Demographic verifiers: | |
| V.2.3.2.1 | Phenotypic shifts show no significant change |
| V.2.3.2.2 | Age/size class shifts show no significant change |

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| V.2.3.2.3 | Environmental shifts show no significant change |
| Genetic verifiers: | |
| V.2.3.2.4 | Genotypic frequency shifts show no significant change |
| V.2.3.2.5 | Marker frequency shifts show no significant change |
| V.2.3.2.6 | Genetic mean shifts show no significant change |
| I.2.3.3 | There are no changes in gene flow/migration |
| Demographic verifiers: | |
| V.2.3.3.1 | Physical isolation shows no significant change |
| V.2.3.3.2 | Mating isolation shows no significant change |
| V.2.3.3.3 | Seed dispersal shows no significant change |
| V.2.3.3.4 | Pollen dispersal shows no significant change |
| Genetic verifier: | |
| V.2.3.3.5 | Gene flow shows no significant change |
| I.2.3.4 | There are no changes in the mating system |
| Demographic verifiers: | |
| V.2.3.4.1 | Parental pool size shows no significant change |
| V.2.3.4.2 | Seed germination shows no significant change |
| V.2.3.4.3 | Pollinator abundance is maintained |
| V.2.3.4.4 | Sex ratio is maintained |
| Genetic verifiers: | |
| V.2.3.4.5 | Out-crossing rate shows no significant change |
| V.2.3.4.6 | Correlated mating shows no significant change |