Biodversity conservation and non-governmental organisations in Oaxaca, Mexico

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Abstract

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The lack of local scale biodiversity assessment in Oaxacan conservation is examined. Biodiversity assessment is a prerequisite of systematic, scientifically directed conservation and in Oaxaca, as in many other parts of the world, conservation is not planned according to scientific prescriptions. This thesis investigates the reasons for this in two ways. First, it considers the technical demands of biodiversity assessment from the point of view of local conservation NGOs. Second, it considers the institutional context in which the concept of biodiversity is translated from scientific discourses to Oaxacan NGOs.

It is argued that tree diversity assessment techniques as currently promoted in scientific discourses are not necessarily appropriate to the needs of local NGOs and that biodiversity is itself a contested concept in Oaxaca. This results in the lack of priority given by Oaxaca's local conservation NGOs to biodiversity assessment. It is further shown that non-systematic conservation has made an important contribution to biodiversity conservation in Oaxaca, and it is argued that it is unrealistic to expect scientific prescriptions for biodiversity planning to be translated, without modification, to rural Oaxaca.

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Declaration

The contents of this thesis and the research upon which it is based are the work of the author alone and have not been submitted for examination elsewhere.

Dedication

This thesis is dedicated to my parents, Mary Ann and David, and to my wife Cristina for all their support, past present and future, and to David Mabberley who introduced me to tropical biodiversity and showed me the need for philosophy in science.

List of Tables and Figures

List of Abbreviations and Acronyms

CBD	Convention on Biological Diversity
CI	Conservation International
CIE	Crude Inventory Efficiency
CIIDIR	Centro de Investigación Interdisciplinaria de Desarrollo Integrado Rural (Centre for Interdisciplinary Research on Integrated Rural Development)
CONABIO	Comisión Nacional para el Uso y el Conocimiento de la Biodiversidad (The National Commission for the Use and Understanding of Biodiversity)
CONACYT	Consejo Nacional de Ciencia y Tecnología (The National Council for Science and Technology)
CONANP	Comisión Nacional para las Areas Naturales Protegidas (The National Commission for Natural Protected Areas)
dbh	diameter at breast height
DFID	Department for International Development
FMCN	Fondo Mexicano para la Conservación de la Naturaleza (Mexican Fund for Nature Conservation)
FONATUR	Fondo Nacional del Turismo (National Tourism Fund)
GEF	Global Environment Facility
GIS	Geographic Information Systems
ha	hectare
IEE	Instituto Estatal de la Ecología (State Ecology Institute)
INBio	Instituto Nacional de la Biodiversidad (National Institute of Biodiversity)
INEGI	Instituto Nacional de Estadística Geográfica e Informática (National Institute Geographical Statistics and Information)
ITAO	Instituto Tecnológico Agrícola de Oaxaca (Technological Institute of Agriculture Oaxaca)
IUCN	World Conservation Union
m.a.s.l.	metres above sea level
MEXU	National Herbarium, Mexico
NAS	National Academy of Sciences
NGO	Non-Governmental Organisation
PRI	Partido de Revolución Instituticional (Institutional Revolution Party)
RBA	Rapid Botanical Assessment
SCAP	Sistema Comunitaría de Áreas Protegidas (Communal System of Protected Areas)
SEDESOL	Secretaría de Desarrollo Social (The Ministry of Social Development)

SEMARNAT	Secretaría del Medio Ambiente y Recursos Naturales (The Ministry of Environment and Natural Resources)
SICOBI	Sistema Comunitario para la Biodiversidad (Community System for Biodiversidad)
UNDP	United Nations Development Program
UNEP	United Nations Environment Program
WWF	World Wide Fund for Nature

Chapter 1: Introduction.

1.1 Antecedents: The Gap Between Theory and Practice in Nature Conservation

Between 1997 and 2000 I co-ordinated a UK Government funded research project which evaluated the potential to conserve the tree diversity of Mesoamerican dry forests in the agricultural landscapes of Oaxaca, Mexico and Honduras. Oaxaca was chosen as one of the case study areas because it had been identified by various biologists as an area of important biodiversity (Ceballos *et al.* 1998; Lorence & García Mendoza 1989) and one which contains considerable areas of tropical dry forest, particularly on the coastal plain. Tropical dry forests, closed canopy forests that usually respond to seasonal rainfall patterns by losing their leaves during the dry season, are considered to be one of the most endangered of tropical ecosystems (Janzen 1988; Lerdau *et al.* 1991). Besides tropical dry forests, several other high diversity and threatened ecosystems have made Oaxaca an appropriate place to practice biodiversity conservation and consequently the state has attracted financial resources from a number of national and international agencies, as will be described later in the thesis. In turn, this had been partially responsible for stimulating the growth of a diverse group of non-governmental organisations based in Oaxaca (hereafter referred to as local NGOs) that work for the conservation of Oaxaca's diverse natural ecosystems. These local NGOs are the central theme of this thesis.

Despite the scientific and financial resources that had been bought to bear on Oaxaca, it became obvious during the course of that project that many of the NGOs working there did not have a clear understanding of the biodiversity of the forests they hoped to conserve. The most obvious manifestation of this was a lack of biodiversity assessment, that is, the surveying of sites and the comparison of their species assemblages. This omission prevents the kind of 'rational' or 'systematic' conservation planning (Pressey *et al.* 1993) that biodiversity scientists argue is necessary in the face of multiple threats to biodiversity and limited resources. Such planning, it is argued, directs funding to sites and areas with especially high numbers of species, or high numbers of threatened species, thus resulting in more efficient conservation. Given the resources available for conservation in Oaxaca, scientific interest in the biodiversity of Mexico and the influence of biologists on the politics of conservation, this gap between science and practice would seem to be a surprising. , and one which is not confined to Oaxaca (Griffiths 2004; Prendergast *et al.* 1999; Sutherland *et al.* 2004). This thesis investigates this paradox.

One approach to resolving this paradox is to approach it as a technological question. Biodiversity assessment is technically demanding, particularly in the tropics where forests are often very diverse. Assessment usually requires the identification of a large number of species and the sampling protocols used require statistical analysis. Together these may form considerable technical barriers for the local NGOs that might otherwise wish to assess Oaxacan biodiversity. Hence part of this thesis considers the technical merits of biodiversity assessment, how it is currently performed by scientists, whether some techniques are better than others and what biodiversity assessment tells us about current conservation in Oaxacan dry forest.

A second approach to explaining this paradox is to ask whether the concept of biodiversity is understood by Oaxaca's NGO community in the same ways that it is understood by biodiversity scientists. If it is not, then a lack of biodiversity assessment, and a consequent lack of uptake of prescriptions for systematic conservation planning, would be understandable. Hence this thesis also considers the social context of biodiversity assessment in Oaxaca, the origins of the term biodiversity, how that term is translated and contextualised by Oaxacan NGOs, and the politics of the engagement of local NGOs and their national and international partners and funders.

The investigation described here therefore required multidisciplinary research that used the techniques and traditions of the biological and the social sciences in order to approach an understanding of conservation in Oaxaca, Mexico.

1.2 Positionality

The formulation of this research is very much a product of my background and experience as a biodiversity scientist working in conservation in Mexico and Central America. My understanding of the concept of biodiversity comes from a background in northern educational and research institutes and it is a concept that I have come to find informative and useful. Furthermore, I am, at least to some extent, a proponent of what here will be called the *biodiversity agenda*, as I am convinced that reducing the rate of loss of biodiversity will be beneficial to humanity. The first approach described above, that of technical deficiency, is a logical consequence of that positionality. It proceeds as if it were reasonable to 'practice' biodiversity conservation in Oaxaca despite the cultural, economic and political divide that separates me from Southern Mexico.

This second approach, that biodiversity and its conservation may be contestable concepts, emerges from the same work in Mexico and Central America but is more problematic in that it potentially represents a conflict with my positionality by questioning the validity of the first approach. This was dealt with by maintaining an epistemic separation between the two approaches that is reflected in the separation of themes in the following chapters, at least until the final discussion.

There was also the potential for problems to be encountered in the process of collecting information through interviews, the main research methodology employed for the socio-economic part of the investigation. For example, there was an obvious possibility that interviewees would give me the answers that they would expect an outsider/biodiversity scientist to want. These tendencies were minimized by assurances of anonymity, through appropriate forms of questioning and through triangulation. Triangulation was provided by interviewing individuals involved in Oaxacan conservation but from outside the community of NGOs and by a brief interview survey conducted in the neighbouring state of Chiapas. Ultimately, however, I could not escape my positionality and whilst the research here described raises questions about the translation of the concept of biodiversity and the politics of its conservation in Oaxaca, it does not question the fundamental justification of biodiversity conservation, i.e. that the loss of species diversity is a bad thing.

1.3 The Structure of the Thesis

The nine chapters that follow begin with a literature review followed by a short chapter describing the geography of Oaxaca. Then follow six chapters that each address specific questions based on primary data. Each is written as a stand-alone 'article' reaching its own conclusions. This requires a certain amount of repetition in their introductory sections, although this is kept to a minimum. Consequently methodological issues are dealt with in each chapter as appropriate, rather than through a single methodology chapter. Finally a concluding chapter draws together the separate conclusions and makes recommendations.

The following preview is intended to provide the reader with a map of the logic of the relationships between each chapter.

Chapter 2. Literature Review

The literature review does not attempt to review all literature relevant to biodiversity assessment and the politics of modern nature conservation, a task that could not be adequately completed within the confines of a doctoral thesis. Instead, it attempts to show that there is an international agenda for the conservation of biodiversity that is situated in the North[1], from where the concept of *nature as biodiversity* comes. This biodiversity agenda, as I call it, has its origins in an epistemic community of scientists and institutions distanciated from the sites of critical biodiversity that are found in the South. The argument that biodiversity, and therefore biodiversity conservation, are situated, and therefore cannot be assumed to be universally understood, is important in that it suggests that biodiversity may be a contestable concept and consequently contextualises much of the work that follows.

Chapter 3. The Geography of Oaxaca and Oaxacan Dry Forest.

This short chapter provides relevant background information. It summarises the current and historic physical, social and economic geography of Oaxaca and the biogeography of Mexican dry forest. It does so with particular reference to the state capital, Oaxaca de Júarez and the coastal municipality of Santa María Huatulco, the two geographic foci of the research described.

Chapter 4. The Practice of Biodiversity Assessment in a Tropical Forest in Mexico.

This chapter explores the processes by which biodiversity assessment becomes authoritative, and does so from the perspective of the practice of biodiversity assessment. It is based on participant-observation of a tree diversity assessment carried out in the tropical dry forests of Oaxaca. Borrowing from Bruno Latour it is suggested that in doing an assessment a forest is redefined through the

[1] Throughout this thesis 'the North' is used as short-hand for the higher income, 'developed' countries, the South is used for the middle and lower income, 'developing' countries.

performance of immutable mobiles that circulate in the networks of a global epistemic community of biodiversity scientists and policy makers. It is argued that scientists produce their accounts of forests at the expense of local accounts whilst, paradoxically, being dependent upon local knowledge for those assessments.

Chapter 5: Assessing Woody Diversity in a Tropical Forest- Local Variation and its Effect on Regional Scale Assessments

This chapter evaluates biodiversity assessment on its scientific merits. It evaluates a methodology for the inventory of plant diversity developed by Alwyn Gentry that has since been applied widely by biodiversity scientists. It tests whether sampling a single site from a locality, as has typically been carried out in regional rapid biodiversity assessments, results in adequate representation of the diversity of that locality for the purposes of comparison with other localities elsewhere in the region or world. This question is addressed with reference to assessments of tropical dry forest from elsewhere in Mesoamerica. Consideration is given to overall diversity as well as diversity of restricted range species.

Chapter 6: Techniques for Efficient Floristic Inventory

The aim of this chapter is to test methods for the rapid inventory of tropical forest tree and shrub diversity in eight seasonally dry tropical forests sites in southern Mexico. It compares fixed area methods (including Alwyn Gentry's methodology) with fixed count and *ad hoc* methods in the context of the needs of financially and technically limited local NGOs. Here the sole objective of rapid inventory is assumed to be the systematic prioritisation of sites for conservation. The efficiency of each method is estimated and compared and the results of site prioritisation, by species number and by complementarity, are compared under the assumptions that the target of conservation should be: a) to conserve maximum species richness; b) to conserve the maximum number of threatened species.

Chapter 7. Biodiversity Networks in Oaxaca

Here an attempt is made to trace the spread of the biodiversity agenda in general, and the concept of biodiversity in particular, through a description of the institutional actors involved in Oaxacan conservation. In so doing it identifies the power relations within a community of local and international non-governmental organisations that fund and implement conservation in Oaxaca. It shows how the state has restricted its role, allowing international and local NGOs to become dominant actors, at least outside of the national park system, with the local NGOs combining agrarian development with biodiversity conservation. This institutional arrangement significantly constrains the translation of scientific prescriptions for nature conservation.

Chapter 8: Partnership and Landscape in Oaxacan Conservation

In this chapter consideration is given to how the community of local NGOs in Oaxaca interpret the 'northern' concept of biodiversity and how it is reconfigured in their work. Opinions and views given by members of that community on the meaning and translation of biodiversity during an interview survey are described and discussed. The preferred scientific configuration of biodiversity as *species richness* is contrasted with an emerging reconfiguration of biodiversity as a function of variability in land use practices. That this central concept is reconfigured in such a way raises questions as to the meaning of partnership between these NGOs and their national and international funding partners. It is argued that Oaxaca's local conservation NGOs are not submissive to these partners, but are capable of contesting the received discourses of international conservation.

Chapter 9: An Assessment of ad hoc *Reserve Selection in Oaxaca's dry forests.*

Here an existing reserve network that was designed without recourse to biodiversity assessment is subjected to assessment to measure its efficiency as defined by biodiversity scientists. The analysis considers whether the reserve network includes the most diverse sites, as measured by three diversity indices, and how its efficiency in representing the local tree flora compares to a network designed by a selection algorithm. This is done by comparing rapid biodiversity assessments of the tree flora of each of the reserves with similar assessments from neighbouring unprotected forests. Thus a GAP analysis is performed in which the degree to which these currently unprotected forests could contribute to the overall list of protected species is assessed. The results are then compared to a similar analysis focusing on presence of threatened species.

Chapter 10: Conclusions

This chapter reviews and concludes the thesis.

1.4 Aim of the Thesis

The aim of the thesis is therefore to consider the influence of the concept of biodiversity on Oaxacan conservation in general and on Oaxaca's NGOs in particular and to describe and critique the ways in the concept is manipulated by those NGOs and the scientists that attempt to assess Oaxaca's biodiversity.

1.5 Summary of Research Questions Addressed

How do local/vernacular representations of nature compete with the scientific construct of *nature as biodiversity* in the context of the assessment of Oaxacan dry forest diversity? (Chapter 4).

Does a currently popular tree diversity assessment technique adequately capture local (within-locality) diversity when used to compare regional (between-locality) diversity? (Chapter 5).

For Mexican tropical dry forests, what are the most efficient methods of assessing tree diversity for the systematic selection of sites for conservation? (Chapter 6).

How are the institutional networks of conservation in Oaxaca constructed and how does this construction constrain the translation of scientific prescriptions for biodiversity conservation? (Chapter 7).

How do Oaxaca's local NGOs construct biodiversity and how does this construction influence the politics of partnership between them and their national and international funders? (Chapter 8).

How does a real, non-systematically designed dry forest reserve system in Oaxaca compare to a hypothetical but systematically designed reserve system for the same forests? (Chapter 9).

Chapter 2: The Biodiversity Agenda and Mexican NGOs- A literature review.

> Just as all social arrangements, including those of communication, involve relatedness to power, so also it is true of ideas [....] ideas and idea systems are often monopolized by power groups and rendered self enclosed and self referential. (Wolf 1999)

2.1 The Construction of Biodiversity

Arturo Escobar (1999) proposes an anti-essentialistic view of nature, arguing that nature exists as a set of socio-culturally informed constructions that articulate and compete side by side. What matters are the ways in which these differently situated views of nature 'vie for control of the social and the biological' (ibid p. 5). As a constructivist view this is not without its critics, as Stonich (1999) points out, natural phenomena may be perceived in very similar ways across cultural and historic divides and thus extreme constructivism may overstate the degree to which humans make nature at the expense of the degree to which nature makes humans. Whilst accepting this criticism, it remains true that no one culture, belief system or institution can claim to know nature better than any other does. The biologist's construction of nature is no more or less valid than that of the farmer or of the shaman and, just as importantly, no view should necessarily be expected to displace another over time or space. Even within particular social groupings, nature is plural and hybrid; it is experienced and valued by different people in different ways. This is especially relevant to environmental problems that are defined on a global scale but that are addressed locally under different cultural circumstances. This is not always acknowledged by those who set out to confront environmental problems from the starting point of a single conceptualisation of nature.

This review seeks to trace the development of a modern construction of nature, that of *nature as biodiversity*, from a starting point in North America and follow its progress to a particular cultural interface, that between the donor agencies/international non-governmental organisations (NGOs) and the local NGOs of Mexico.

In 1986 the National Academy of Sciences (NAS) and the Smithsonian Institute sponsored *The National Forum on Biodiversity* in Washington D.C. Contributions were made at this scientific meeting by those considered to be amongst the most eminent and influential in the field of conservation biology. Previously the term *biological diversity* had been used but the contraction, *biodiversity*, was coined for this forum (Wilson 1997) and used again in the remarkably popular proceedings, *BioDiversity*, that followed (Wilson 1988). As a result, the term *biodiversity* has, with surprising rapidity, entered into scientific, popular and political discourse. Whilst it is not claimed that modern biodiversity conservation began at this NAS forum, this meeting did represent an important milestone in environmental thinking and serves as a useful point of departure for this review.

What is revealing about the proceedings is that not only are they derived from a forum held in an industrialised northern nation but that of the 61 'biodiversity scientists' who made published contributions, 60 were representatives of institutions based in the North, whilst just one was a representative of an institution from the lower income countries. This is despite the recognition given by many of the authors to the global nature of the problem of biodiversity loss and the preponderance of that biodiversity in the tropics of the developing South.

The science of conservation biology has grown out of the ecological sciences alongside, and in response to, an increasing concern for the environment and the damaging effects of human activities on it. Takacs (1996) traces this concern, as it has influenced modern biodiversity scientists, through northern thinkers including Aldo Leopold, Rachel Carson and Paul Ehrenfeld. Each of these, in their different ways, formulated a vision of a human-made environmental crisis and as a result became strong advocates for the conservation of nature, a goal that conservation biology has taken up. Conservation biology therefore is a goal-directed science and it is inevitable that biodiversity scientists are, almost without exception, supporters of that goal. Many biodiversity scientists have taken to the role of advocate for conservation with zeal and have not only tried to measure, model and explain biodiversity and its loss, but have also been proactive supporters of the politics of its protection (Ehrlich 1988; Janzen 1994; Mittermeier *et al.* 1998; Pimm *et al.* 2001; Terborgh 2000). Thus, biodiversity as a concept has become intimately associated with the politics of environmentalism and, more specifically, the politics of what is known as the biodiversity crisis, forming an agenda for biodiversity conservation.

This conflation of science and political advocacy is here referred to as the *biodiversity agenda.*

Biodiversity advocates need more than quantifiable evidence of nature's disappearance to justify conservation. They also need convincing reasons as to why we should want to safeguard the world's patrimony of species and ecosystems. Biodiversity scientists, having grasped the advocate role, have also been at the vanguard of those providing those justifications. A typical example is that of Ehrlich and Ehrlich (1992) in which they formulate their justifications under four types of value: *ethical*, *esthetic*, *direct economic* and *indirect economic,* each of which is diminished by the loss of biodiversity. The first two values are highly culturally specific, and thus it cannot be assumed that those outside of northern research institutions share them. The second two are also problematic, but in a different way, in that they are values that will not necessarily accrue to all members of all societies equally, or more important still, to those who are asked to forego other 'values' in order to preserve biodiversity. As McAfee (1999) argues, the utilitarian concern for future global welfare may in practice turn out to be primarily about the welfare of the rich industrialized nations. The point here is that these justifications are just as constructed and situated as is the *nature as biodiversity* conceptualisation to which they relate. Of course, nature, and rationales for its conservation, can be conceptualised in very different cultural contexts and with equal validity, it is not a problem only recognisable or approachable via the northern biodiversity agenda. Examples include Agwan (1999) who describes a reverence of biodiversity inherent in Islamic teachings, whilst Allendorf (1997) and Regosin and Frankel (2000) approach conservation issues from Zen Buddhist and Jewish perspectives respectively. However, biodiversity scientists and the perspectives they bring to nature conservation have come to be dominant. As Takacs (1996 p. 99) summarises:

> The term biodiversity makes concrete- and promotes action on behalf of- a way of being, a way of thinking, a way of feeling, and a way of perceiving the world. It encompasses the multiplicity of scientists' factual, political and emotional arguments in defence of nature whilst simultaneously appearing as a purely scientific, objective entity'. [...] simultaneously they create our worries and pose themselves as palliatives to those worries. And their answers always require us to pay more attention to their own expertise and save more of what they want to save- that is, as much biodiversity as possible.

Whilst coming to dominance in worldwide environmental politics, discourses surrounding biodiversity have remained situated in the developed North. Eleven years after the publication of *BioDiversity*, a follow up volume of invited papers, *Biodiversity II*, was published (Reaka-Kudla *et al.* 1997) the purpose of which was to chart progress in biodiversity conservation since the NAS forum. Its 50 contributors include 49 representatives of US institutions and one from Australia. The degree to which northern institutions have continued to dominate the discourse on biodiversity in these two publications is indicative of the degree to which the biodiversity agenda is situated in northern scientific discourses.

Biodiversity is a quantitative conceptualisation of nature as is encapsulated by the title of Kevin Gaston's book *Biodiversity: a biology of numbers and difference* (1996a). Much of the scientific literature on biodiversity conservation is thus highly technical, concerned as it is with estimating species numbers, monitoring changes in those numbers, and measuring the flows and frequencies of genes and organisms within populations. One end for much of this work is the identification of sites where biodiversity conservation should be put into practice. Such sites are often referred to as 'hotspots' (Myers 1988; Reid 1998). The term 'hotspot', like biodiversity before it, has escaped the confines of academic literature to enjoy life in popular and political discourse. These sites are prioritised because of their high numbers of species and/or their high concentrations of threatened species. The effect of this numerical technification is to have further entrenched the discursive power of the biodiversity agenda in northern research institutions. In contrast, the hotspots that the discourse identifies are primarily in the tropics of the south (Myers *et al.* 2000). Biodiversity hotspots have come to be places in the South where institutions dominated by the North think conservation *ought* to happen.

The high standing and respect afforded scientists, especially those seen as motivated by the noble goal of biodiversity conservation, has meant little questioning of the growing hegemony of the biodiversity agenda has taken place (but see below). Rarely either have the numbers that add up to the frightening estimates of species loss and habitat destruction been questioned. As the number of species predicted to be lost varies widely, so too does the potential value forgone by their loss and with it the justification for so much investment in biodiversity conservation. The numeric technification of nature conservation is therefore part of the politics of environmentalism and lends rhetorical advantage to biodiversity scientists. Whilst few disagree that the world faces previously unparalleled rates of species loss, recently some of the more extravagant estimates have been questioned (Lomborg 2001).

Confidence in the 'correctness' of the biodiversity scientists' view of nature and how to go about its conservation should be tempered not only by Escobar's previously noted philosophical arguments about its cultural situatedness but also by a sobering analysis of an earlier *cause célèbre* of conservation- the preservation of wilderness. Gómez-Pompa and Kaus (1992) show that the idea of wilderness, particularly in the US, is not one that easily transfers to different cultural contexts where, for example, wilderness as a place defined by lack of humans is very hard to either conceptualise or identify (see also Denevan 1992). Further, the scientific paradigms of ecological climax and stability that underpin the wilderness concept have shifted. That shift was partially caused by a new understanding of the dynamics of mature ecosystems, especially those in the tropics (Whitmore 1984) and partially by the recognition that, again particularly in the tropics, conservation had to include humans as actors in the landscape, not as something to be expelled from it. The effect of the wilderness concept and its relationship with the 'National Park' paradigm of natural areas to be maintained free of humanity's intervention is still being exposed and explored today. To many it is a model alien to many cultures on to which it has been transplanted and represents a post-colonial enterprise (Guha 2000; Schwartzman *et al.* 2000; Vidal 2001). Just as many biodiversity scientists now question wilderness preservation so too should we ponder the possibility that the current hegemonic paradigm of biodiversity may also shift.

2.2 Biodiversity and GIS Technologies

The quantification of biodiversity has, during the last ten years, lent itself to display and analysis through Geographic Information Systems. Biodiversity scientists have eagerly taken up new GIS technologies; with perhaps the first widely discussed applications being the Gap Analysis Program (GAP) used for the detection of habitat types under-represented in the conservation systems of the USA (Scott & Csuti 1997; Scott *et al.* 1993). GIS is now firmly established as a tool for conservation planning and management in the North and has been taken up by international conservation organisations based in the North but whose activities are orientated towards conservation in the South (e.g. Redford *et al.* 2003). Critical analysis of the role of GIS, and its effects on power relations between stakeholders in biodiversity conservation has been less thorough compared to its application elsewhere. Schuurman (2000) reviews the geographical literature's critiques of GIS and its potential benefits and disbenefits, whilst Dunn *et al.* (1997) consider the promotion of GIS in relation to the circumstances of developing countries. Much of the debate contained within these articles is of relevance to GIS and biodiversity conservation in the tropics of the developing South, especially considerations of the discursive power that GIS lends to those who have access to the technology, compared to those who do not, and of the difficulty of incorporating social relations into GIS. Bowker (2000 p. 751) argues that the diverse disciplines that deal with biodiversity are never likely to agree on a standard '*atomisation*' of their common ground. It therefore falls to GIS users to deal with ontological difference:

> In a biodiverse world, we need to be thinking through ways of manipulating ontologically diverse data. It is surely a rich challenge to the GIS community to devise forms of representation which integrate without traducing the multiple data diversities of the field of biodiversity.

Whilst this is a welcome critique his principal consideration is that of the ontological differences within biodiversity science, rather than the differences that lie between the biological, political and social sciences. Biodiversity scientists have paid little attention to the ramifications of the cultural situatedness of these technologies, especially in relation to the resource and technical limitations typically faced by the southern actors expected to put conservation theory into practice. The extent to which the transfer of these technologies can be done appropriately or successfully in biodiversity conservation is only beginning to be explored. Sieber (2000 p. 778) in discussing 'grass roots' conservation organisations in USA, highlights the power relations involved in such technologies:

> By transforming [themselves] the GIS groups can change the technology that is used by the power elites. This is perhaps easier in the conservation movement, where conservation scientists, activists, and GIS developers move regularly between roles [...] ownership of GIS is considered to be essential to successful adoption.

Martin (2000) gives us one of few examples of the importance of inter-institutional relationships inherent in the application of GIS by local NGOs working for conservation in lower income countries, in his case from Ecuador, and his analysis highlights the importance of continued donor support in the maintenance of GIS.

Whilst GIS is now a key component of the information technology systems of international NGOs it is not clear to what degree its transfer to local NGOs is sustainable or beneficial.

2.3 Scientists, International NGOs and Interstate Agreements

The biodiversity scientists' view of nature, and all the baggage carried with it, might be of only passing interest if it were enclosed entirely within the discipline, however, it is not. Instead it must be seen in the context of the influence scientists have within international NGO politics, as will be discussed below, and more generally over environmental policy formulation. Raustiala (1997a p. 487) suggests this may be because:

> [T]he scientific nature of environmental problems and the authoritative position of expert communities generally grant experts considerable influence over policy.

Raustiala (1997b) argues that international NGOs are increasingly important participants in international environmental institutions. This he interprets as being beneficial not only because it strengthens the voice of civil society in conservation, but also because it is an aid to states in their role as global regulators. Certainly, the association between those transnational institutions with an interest in biodiversity conservation[2] and international NGOs is intimate as is demonstrated by collaborations between World Wide Fund for Nature (WWF) and The World Bank (e.g. Dinerstein *et al.* 1995). In turn, international NGOs depend heavily on biodiversity scientists in the setting of their priorities and in the legitimisation of their advocacy (da Fonseca 2003; Redford *et al.* 2003). International NGOs such as WWF, The World Conservation Union (IUCN), Conservation International and Birdlife International are all now involved in the highest levels of policy formulation and include amongst their staff many of the internationally respected biodiversity scientists who have been instrumental in setting the biodiversity agenda. Thus the international community of NGOs working for conservation and the community of biodiversity scientists are not easily separable but merge to form a powerful epistemic community. Here an epistemic community is defined as:

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[2] World Bank, UNDP, UNEP

> A network of professionals with recognized expertise and competence in a particular domain and an authoritative claim to policy-relevant knowledge within that domain or issue-area. (Haas 1992 p. 3)

The significance of such a community, in the context of biodiversity conservation, is characterized by Raustiala (1997a p. 487) thus:

> Acting transnationally, a knowledge-based community can create convergence around its preferred policy solutions. Influence in the epistemic model is thus both cognitive and bureaucratic; while epistemic communities help to shape state preferences for cooperation through the knowledge they possess, they also exert influence through the institutionalization of community members into policy-making bureaucracies.

Thapa (1998) describes an early example of how the biodiversity community came to influence international agreements. He reports that the eminent biodiversity scientist, and now principal advisor on biodiversity to the World Bank, Thomas Lovejoy first suggested debt-for-nature swaps in *The New York Times* in 1984. By 1987 the first swap had been mediated by the Washington based NGO Conservation International to alleviate the Bolivian government's debt in return for investment into the Beni Biosphere Reserve.

Perhaps the most impressive achievement of the biodiversity community was the influence their advocacy had on the UNEP Convention on Biological Diversity (CBD). WWF, IUCN and World Resources Institute were instrumental in the formulation of this agreement, which more than any other put the North's biodiversity agenda at the centre stage of world politics. It would be difficult to overstate the importance of this agreement, and Heywood and Iriondo (2003 p. 323) describe its impact thus:

> At a stroke, conservation ceased to be an optional extra and became official, global and national policy.

Thus the discursive hegemony of the biodiversity scientists, via the advocacy of international NGOs, appears to be approaching a political hegemony in international agreements relating to nature conservation.

Raustiala (1997a p. 487) notes that the influence of the community was not entirely a result of objective and dispassionate consideration on the part of the states involved in negotiation of CBD, but instead:

> Variance in state choices reflect[ed] differential access by the community of experts rather than differential receptions of the content of the community's proffered policy solution.

Wapner (1995) also provides a very positive interpretation of the ability of international environmental NGOs, (including WWF) to influence not only governments but also other sectors of civil society and argues that this is evidence of the emergence of a world civil society. However Clark *et al.* (1998) are more sceptical about the extent to which governments allow NGOs to influence international decision making, pointing out that there are clear limits beyond which governments will not allow world civil society to have direct influence. Song and M'Gonigle (2001 p. 386) are also cautious. They consider the political economy in which biodiversity scientists operate and conclude that despite advances:

> Much writing on ecosystem management focuses on the science but not the political and economic processes within which such management operates.

In their estimation biodiversity scientists at intergovernmental fora only address the proximate rather than ultimate causes of environmental problems and that ultimately conservation goals are marginalized because the growth needs of industrial capital are always given a higher priority.

2.4 The Biodiversity Community and Biodiversity Conservation in the South

Despite the reservations of Clark *et al.* (1998) and Song and M'Gonigle (2001) biodiversity scientists, international NGOs and the epistemic community they form continue to have great influence over the discursive and financial aspects of global conservation. The way this influence is felt in the South, where the biodiversity crisis is at its most acute, is crucial to our understanding of the institutional politics of biodiversity conservation.

Guha (2000) questions the right of institutions based in the North to promote nature conservation in the South, given the historical association between conservation and colonialism and the inability of unsustainable, industrial based consumer societies to contain their excessive claims to the South's environmental space. He goes on to conclude that Western environmentalists and their missionary tendencies may be more dangerous and even more hypocritical than those who have tried to impose religious and economic models on developing countries.

In a more nuanced reading based on a case study of biodiversity conservation in Guinea, Fairhead and Leach (2002 p. 109) show how people, science, institutions and politics coalesce to produce adjudications of indigenous relations with nature based on alien concepts that 'reproduce western, colonial distinctions' between nature and culture.

At the level of policy formulation, and contrary to Wapner (1995), Nelson (1997 p. 467) is critical of the role of international conservation NGOs and their influence on the south concluding that:

> Neither the World Bank's structure and mandate nor the NGOs' strategies encourage such a consensus-based process. NGO environmental advocacy has instead tended to broaden the World Bank's authority to regulate its borrowers' economic and environmental policies.

At the level of project and programme implementation Jepson and Canney (2001 p. 226) argue that the 'hotspot' approach strongly championed by the biodiversity community represents a response to a global problem and as a consequence is inherently flawed in its relation to local needs:

> As hotspots are predominantly tropical forest landscapes [...] under threat, successful conservation of these areas will benefit global ecosystem service (e.g. climate regulation). However, the services humans require at the local scale are ignored.

Sheil (2001 p. 1180) notes another weakness in project implementation resulting from the imbalance between northern and southern institutions in discursive power. He is critical of the emphasis biodiversity scientists and donor organisations (which would include international NGOs that fund local conservation initiatives) put on research when tropical conservation projects so often require more direct and practical support:

> Many projects continue to emphasize variables irrelevant to daily management and, even worse, to institute activities that draw precious staff and financial resources away from more critical actions.

Thus it can be concluded that at both policy and implementation levels, the hegemony of the biodiversity community does not necessarily translate into equitable or efficient conservation.

2.5 International NGOs and Local NGOs: an Interface Between the Biodiversity Agenda and Tropical Conservation

Trends in nature conservation

One of the most recently created and financially powerful institutions promoting international biodiversity conservation is the Global Environment Facility (GEF) whose implementing agencies are UNEP, UNDP and the World Bank. In a policy document on public involvement in its projects (GEF 1996) the intention of this institution to involve NGOs in the entire project cycle is made clear:

> Effective public involvement is critical to the success of GEF-financed projects. When done appropriately, public involvement improves the performance and impact of projects by: [...] making use of skills, experiences, and knowledge, in particular, of non-governmental organizations (NGOs), community and local groups, and the private sector in the design, implementation, and evaluation of project activities.

Partnership between international funding organisations and local NGOs is now a common institutional arrangement in the practice of biodiversity conservation across the South (e.g. Bryant 2002; Murombedzi 1999; Taber *et al.* 1997). This is, however, only one of several trends that are detectable in current conservation policy and practice around the world. Wilderness preservation is giving way to an approach that is more sympathetic to human influences. One manifestation of this is that protected areas ('parks') are commonly conceptualised and managed in relation to, rather than separate from, the more anthropogenically altered landscapes in which they are embedded (Bennett 1999; Burke 2000). A related trend is the increased interest in the potential for biodiversity conservation to be practiced outside of protected areas (Boshier *et al.* 2004; Halladay & Gilmour 1995; Pagiola *et al.* 1997). Accepting human activity as an intrinsic, and possibly positive, part of landscapes and their conservation allows for a better integration of local needs and a greater role for local NGOs that represent them.

Hulme and Murphree (1999) describe a tendency towards neoliberalism within biodiversity conservation, in particular the introduction of market forces as a mechanism, highly criticised by McAfee (1999), for directing decision making and implementation. To what extent local NGOs are being co-opted, willingly or otherwise, into this process is not clear but it has strong adherents from within the biodiversity community, being a logical extension of the 'direct economic value' justification for biodiversity conservation (Ehrlich & Ehrlich 1992). Perhaps the best known example of using market forces to fund conservation activities is the bioprospecting agreement between Costa Rica's National Institute for Biodiversity (INBio) and Merck Farmaceutical (Lobo 1994; Sittenfeld *et al.* 1999).

North-South NGO relations in biodiversity conservation.

The northern biodiversity agenda has various interfaces with the middle and lower income countries of the South but one of the most important is that between international NGOs and their local NGO partners, a theme considered in chapters 7 and 8. In the context of biodiversity conservation, this relationship is little explored. The interfaces between northern institutions and southern NGOs in the broader fields of environment and development are better researched, and of relevance here. The reasons for partnership between these sectors are varied but Korten (1992), writing of the involvement of environmental NGOs in forestry, describes the common perception that local NGOs should have a zeal for addressing needs other parts of society have proven to be ineffective in confronting. He suggests that:

> Because of their independence and relatively small size, [local] NGOs are often able to take more controversial stands, act more quickly and innovate more easily than their larger, more established counterparts.

Van Rooy (2000) is confident that, in the context of development orientated NGO activities, international NGOs have done much good work in fostering southern based local NGOs, so much so that global social justice has advanced to the point at which international NGOs are in danger of redundancy and need to rethink their role. Such an optimistic conclusion to relations between international NGOs and their local partners in the South is contested. There is evidence to suggest that innovative zeal and rapid response, the very qualities that northern donors supposedly seek, may be suppressed as a result of the formation of north-south partnerships. Local NGOs have reported that their donor partners too often impose unnecessarily heavy and stifling bureaucratic burdens on them (Mawdsley *et al.* 2002) and Townsend (1999) notes that the loss of independence that can accompany financial ties to northern donors, can turn local NGOs from innovators into mere service providers.

More generally Aldaba (2000) suggests that local development NGOs are in a state of uncertain transition resulting from decentralization and, particularly in Latin America, positive *per capita* economic growth and declining aid flows. He argues that local development NGOs need to find ways of going 'beyond aid' and becoming financially self-sustaining and thus less dependent on paternalistic donors. Similarly Bebbington (1997) considers Andean local NGOs in the similar context of public sector reform and economic liberalization. He notes that local NGOs have become over dependent upon northern donor NGOs with the resulting institutional forms being determined by external relationships rather than relationships with their local client groups and other local NGOs. He thus interprets the donor-southern partner relationship as one that distorts local development NGO relations with their constituency, the rural poor, and that in turn is partially responsible for the limited impact local development NGOs have had on rural poverty. To what extent international conservation NGOs are distorting the agendas of their southern partners is yet to be determined but is explored in this thesis in the context of the discursive, financial and political strength of the biodiversity agenda.

Mexico

The first environmental NGOs in Mexico arose in the 1960s as middle class urban membership groups (Hernández & Fox 1995). Mexico's status as a 'megadiversity state' (Mittermeier *et al.* 1998) has since attracted considerable interest from international NGOs, particularly in the high diversity areas, a disproportionate number of which are in the southern states of Oaxaca, Guerrero, and Chiapas and the Yucatán Peninsula. At the same time decentralisation, democratisation, structural adjustment and economic crisis have, to a greater or lesser extent, reduced the role of the public sector in Latin America. This does not necessarily mean, as Reilley (1995) points out, that NGOs can be expected to easily fill the vacuum left by the state or that they should be used for social experimentation. Yet undoubtedly northern conservation NGOs see their southern partners as useful partners whose 'resources' include relatively inexpensive logistical expertise, local knowledge and freedom from the bureaucratic burden often associated with the public sector. Much of this may be true, indeed as Townsend (1999) notes the NGO sector in Mexico is heavily staffed by university educated professionals who have escaped public sector contraction and are often willing to accept low wages in return for the ability to make a greater impact on the social issues that interest them. Nonetheless, the Mexican NGO sector is considered weak, relative to other Latin American countries, existing as it does in one of the most centralized states of the region. Hernández and Fox (1995 p. 181) suggest that:

> With few notable exceptions, the presence of NGOs in Mexican politics and social change efforts is remarkably limited [a fact attributable mainly to the] 'omnipresent' state whose role in the provision of basic services has been great- and often used to buy political patronage.

The degree of centralisation is now decreasing following the economic crisis provoked by the crash of the Mexican peso in 1994, and increasing democratisation of national politics. This has left greater space in which the NGO sector can operate. This has continued following the election, in 2000, of the first national president not to represent the Institutional Revolution Party (PRI) for over 70 years.

The role of local NGOs in biodiversity conservation in Latin America and elsewhere has been reported in favourable terms (Jones 1999; Taber *et al.* 1997). Morell (1992) reviewing 'grass roots' NGOs and forest conservation in Central America and Mexico points to some of the reasons why such organisations in the region may be more effective in promoting forest conservation and by extension be good partners for northern donor institutions:

> This community perception of organizational objectives - people and production first - contrasts with the 'usual' approach of government agencies, development organizations and non-local NGOs, whose reasoning is the exact opposite: forest conservation will, in turn, solve community problems.

The implication of this is that the effectiveness of local NGOs in conservation is a result of their unwillingness to accept goals set by outside organisations, their organisational goals therefore being in conflict with their erstwhile partners. With reference to a case study of the political ecology of forest conservation in southern Mexico, Haenn (1999) shows how local organizations can allow competing views of forest conservation ('alternative environmentalism') to co-exist with those of state sponsored conservation. In this case, it might have been concluded that the role of local NGOs was to act as intermediaries between the global biodiversity agenda and local needs.

Given reduced aid flows, international conservation NGOs will continue to be attractive funding partners to a range of local NGOs, including those previously not orientated towards biodiversity conservation. This would extend the influence of the biodiversity agenda, even if it were adapted to competing conservation paradigms. This may already be happening given the large investments being made by the GEF in combined development and conservation activities across quite considerable areas of tropical developing countries. An example of this is the Mesoamerican Biological Corridor programme that aims to integrate biodiversity conservation across a considerable portion of southern Mexico with conservation projects in large areas of Central America. Its appraisal document (World Bank 2000) suggests that consultation with other sectors, including the non-governmental, is to be pursued:

> Informed participation is further ensured through the work of the local corridor councils, whose members actively inform their constituencies (farmers, NGOs, academics, local government)

Thus it can be assumed that the interface between a northern conceptualisation of biodiversity and local NGOs in Mexico will remain and that the politics of the translation of the biodiversity agenda from international to local NGOs will continue to influence rural land use in the tropics for some time to come.

2.6 Summary

The politics of nature conservation is intimately associated with the concept of biodiversity. These politics, the biodiversity agenda, are powerful and working at an international level, set standards and promote technical sophistication in the search for solutions to the biodiversity crisis. It is now common for this agenda to be played out in practice through partnership between international organisations and locally based NGOs. This creates a potentially unequal and prescriptive relationship in which the financial power and political authority of those international organisations subjugates the aims and needs of the local NGOs. How local NGOs respond to these circumstances is one of the themes of this thesis.

Chapter 3: Oaxaca and Oaxacan Dry Forest in Context.

3.1 The Biogeography of Mexican Tropical Dry Forest.

A useful point of departure for interpreting Mesoamerican[3] plant biogeography is the floral interchange occasioned by the formation of the Central American Isthmus that united the continents of North and South America during the late Pliocene approximately 3 million years ago. It is assumed that, until that point, the biotas of these landmasses had had separate evolutionary histories since the break up of Pangaea at the end of the Jurassic period approximately 150 million years previously (Coates 1997). The non-random nature of the floral interchange has resulted in an unequal contribution of neotropical (South American) and Boreal (North American) taxa to the various vegetation types found in modern Mesoamerica. A relatively larger contribution is made to the flora of the temperate forests of the highlands by taxa with affinities to North America than is made to lowland, tropical vegetation types. Evidence for this is found in the predominance of boreal genera such *Quercus* (oak) and *Pinus* (pine) amongst the tree diversity of the highlands of much of Mesoamerica (Galindo-Leal *et al.* 2000; Rzedowski 1981) which contrasts with their near complete absence from lowland forests. The tree floras of the lowland, tropical forests of the region, which include tropical dry forests, have usually been considered to be of principally neotropical affinity and hence have been described as northern extensions of South American floral assemblages. In particular, Mexican moist forests are relatively depauperate compared to their counterparts in Central and South America that are more species diverse (Toledo 1982).

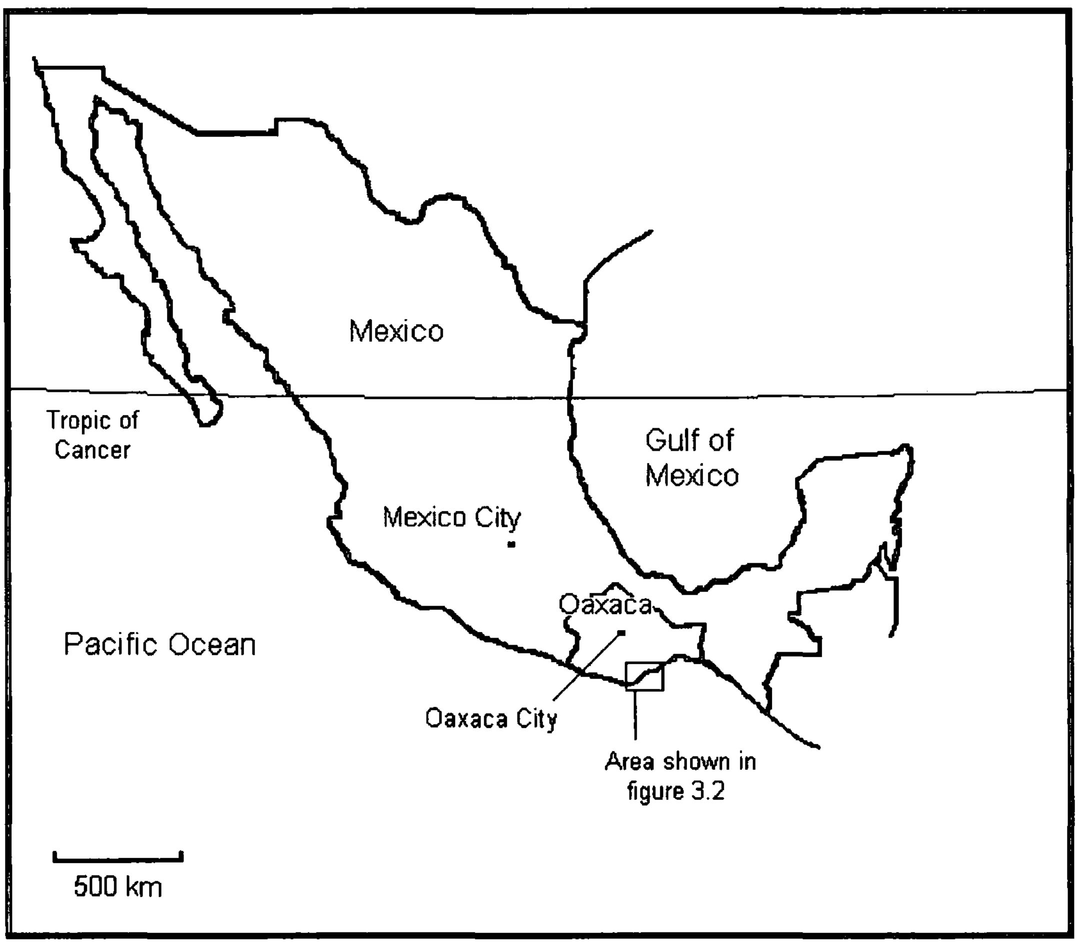
However, this simplistic interpretation has come under revision based on reinterpretation of palaeontological evidence and it is now suggested that a significant proportion of lowland Mesoamerica's tree diversity may have a 'boreotropical' origin that predates the closure of the Isthmus of Central America. It is proposed that interchange may have been possible during a period of geographical proximity between the North American and Eurasian tectonic plates in the late Tertiary (Lavin & Luckow 1993). Taxa of the dry forests of Mesoamerica whose affinities may be boreotropical include one of its most indicative and speciose

[3] Mesoamerica is here defined broadly to include the countries of Central America and Panama and Southern Mexico.

genera, *Bursera*, whose closest generic relatives are now found in Africa, Eurasia being proposed as a land bridge for its migration to North America.

Speciation after the joining of North and South America is also controversial. Pleistocene climatic change, during alternate dry and wet climatic periods occasioned by glaciations, has been proposed as a driver of speciation in North and South American dry forests (Pennington *et al.* 2000). Whilst this 'refuge theory' has been criticised (Mayle 2004), recent evidence suggests that it may still be relevant to the floral biogeography of Mexican dry forests (Lavin & Luckow 1993).

Figure 3.1 Map of Mexico and the state of Oaxaca



It is useful to follow Ceballos (1995) in dividing the Mesoamerican dry forests into three principal phytogeographic areas:

- The Central American dry forests that extend from western Guatemala to northwest Costa Rica and to which should be added the isolated dry forests on the Pacific coast of Panama;
- The dry forests of the north and northwest of the Yucatán Peninsula of Mexico and;
- The western Mexican dry forests, which include those of Oaxaca (figure 3.1), extend north from the Isthmus of Tehuantepec to Sonora (beyond what is usually considered Mesoamerica).

These forests share broad similarities in the composition of their woody diversity, the Leguminosae (particularly *Cassia*, *Lonchocarpus* and *Acacia*) being the dominant family by number of species. Important contributions are also made by Bignoniaceae (particularly amongst the lianas) Euphorbiaceae (especially *Croton*), Rubiaceae, and Boraginaceae (especially *Cordia*). These families are also well represented in the wet tropical forests of the region, leading Gentry (1995) to propose that, at the familial level, dry forest floras are essentially depauperate analogues of wet forests. However, he goes on to note three important exceptions to this, Cactaceae, Capparidaiceae and Zygophyllaceae, families present in dry forest but largely absent from wet forests.

Between these three phytogeographic areas, the strongest floristic affinities appear to be between the Yucatán Peninsula and Central America (Estrada-Loera 1991; Ibarra-Manríquez *et al.* 2002). It is suggested that the swamps of Tabasco, the mountains of Chiapas and the Isthmus of Tehuantepec have been a barrier to floral exchange with the Western Mexican dry forests. Western Mexican dry forests may instead have greater affinity with those of northern South America than with either Central America or the Yucatán Peninsula. Gentry (1995) offers evidence that a greater proportion of dry forest genera are uniquely shared by Chamela in Western Mexico and forests sampled in Venezuela and Colombia than are shared by Chamela and the Central American forests of Guanacaste, Costa Rica.

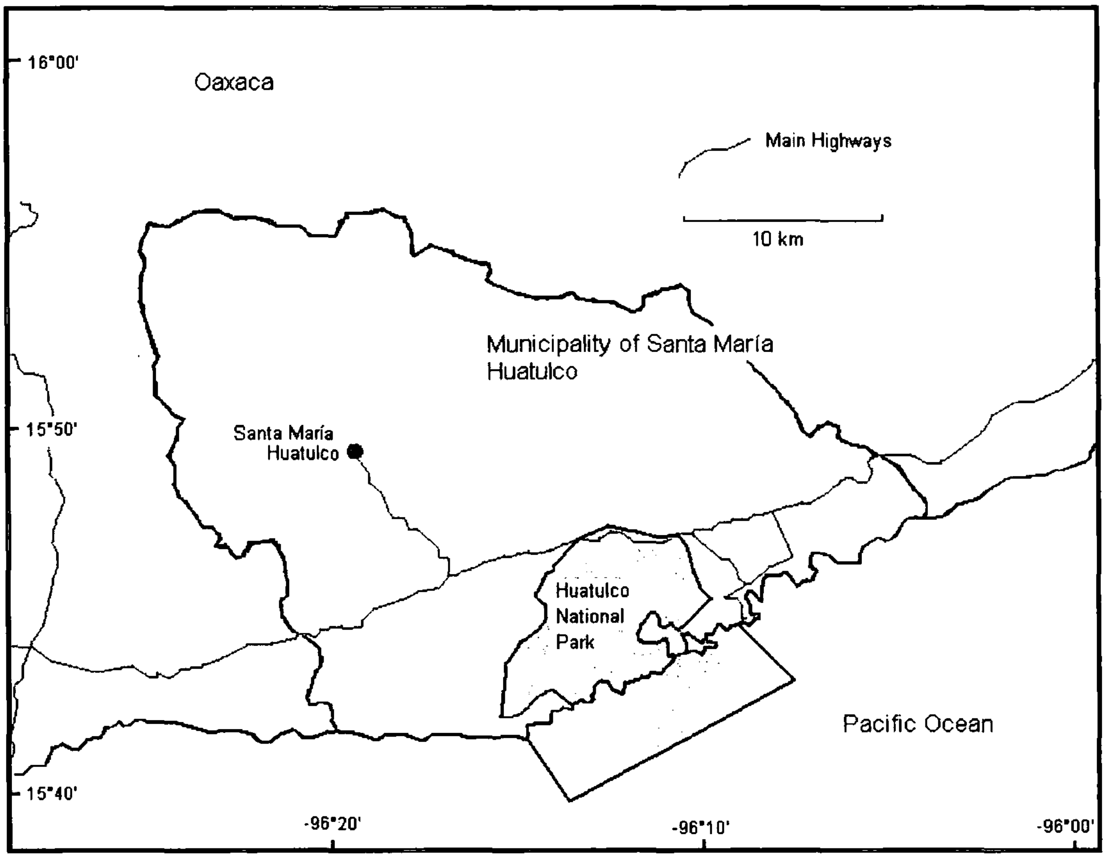
The dry forests of Mesoamerica are highly variable in species richness. Trejo and Dirzo (2002) took 1 ha samples from twenty sites across Mexico and found tree diversity (woody plants with stem diameters > 2.5 cm) to vary from 22 to 97 species per sample (mean = 57). Using a similar protocol Gillespie *et al.* (2000) sampled seven tropical dry forests in Nicaragua and Costa Rica and found tree species richness (stem diameter < 2.5 cm) to vary between 44 and 77 (mean = 56). Trejo and Dirzo 2002 were able show that 76% of this variation was explained by variation in potential evapotranspiration (PET), with high PET (measured by Thornthwaite's Index) correlated with high species richness. They also showed that species turnover between sites was high, with 72% of species being found in only one of their twenty samples and no species being present in all samples. Caution must, however, be advised in relation to these results given that the sampling technique employed is shown to be suspect in chapter 5 of this thesis.

Despite their relatively depauperate floras, at least compared to tropical wet forests, Mesoamerican dry forests are relatively rich in endemics[4] and this is especially so of those of western Mexico (Gentry 1995). Indeed endemism in Mexico appears to increase along a gradient from mesic to xeric habitat types (Challenger 1998) with desert habitats being, proportionately, the richest of all. Allopatric speciation driven by fragmentation and isolation during the Pleistocene climatic changes, noted above, has been proposed as a possible reason for high dry forest endemism. This isolation may have been exacerbated for the forests of Western Mexico that are typically contiguous with the floristically very distinct temperate pine and oak forests and hence isolated from the tropical wet forests from which their flora is thought to be derived (Ceballos 1995). Whatever the reason for high levels of endemism, this characteristic makes the conservation of Oaxacan dry forest a high priority for international conservation.

It should be noted that this division of Mesoamerican dry forest into three blocks of vegetation on the continental coastal fringes obscures the great many, if smaller, areas of dry forest in the interior valleys of the region. These forests have largely been overlooked, (but see Trejo & Dirzo 2002), perhaps because such valleys have often been the preferred sites of settlement since pre-Colombian times with the result that in many places almost no forest remains. Nonetheless, because of their relative isolation we might expect higher levels of endemism in the remaining fragments.

[4] Edemics are species restricted to a specified area, in this case Mesoamerican dry forest.

Figure 3.2 Map of the Municipality of Santa María Huatulco, Oaxaca, Mexico



Much of the municipality of Santa María Huatulco (henceforth referred to as Huatulco, figure 3.2) on the coastal plain of Oaxaca remains under forest cover, albeit fragmented and disturbed forest and forest fallow. Forest fallow forms part of the traditional cycle of land management that provides for subsistence needs in maize, beans and livestock. In the higher areas (above 300 m.a.s.l), in the north of the municipality, shade grown coffee also contributes to forest cover. Below that altitude the natural vegetation type of Huatulco is tropical dry forest and on the relatively poor sandy acidic regosol soils that can occasionally be rocky, especially near to the shoreline. Much of the area is under closed canopy forest, but substantial areas of this are forest fallows, rather than mature forest. However the difference between the two is often not obvious with maturer fallows grading imperceptibly into more intervened forest. On rockier and steeper slopes forest still dominates but can be notably less dense. Perhaps because of the abundance of forest and forest fallows, trees are not always integrated into crop fields or in pasture. Forest is found right up to the coast line, with only occasional and small patches of mangrove found around the bays. The forests and forest fallows of

Huatulco (see figures 3.8 to 3.13) are used as a case study to investigate techniques for tree diversity assessment in this thesis.

3.2 The Physical Geography of Oaxaca

The state of Oaxaca (henceforth referred to as Oaxaca) is located on the southern Pacific coast of Mexico. It has an area of approximately 95 000 km^2 and approximate longitudinal and latitudinal ranges of 16° N to18° N and 99° W to 94° W, respectively. Two mountain ranges (*sierras*) converge in Oaxaca and define its physical geography. The Sierra Madre Oriental (also know as the Sierra de Juárez) runs northwest to southeast across the northeast of the state, the Sierra Madre del Sur runs parallel to the Sierra Madre Oriental across the centre and south of the state. A third sierra, the Sierra Atravesada, represents the most north-westerly extension of the Sierra Madre de Chiapas that enters the east of the state along the Isthmus of Tehuantepec.

Within Oaxaca, the geographical foci of this thesis are:

1. The state capital, La Ciudad de Oaxaca de Juárez (henceforth referred to as Oaxaca City) located at approximately 1500 m.a.s.l. at the intersection of the three flat fertile valleys (the *Valles Centrales*) which nestle between the Sierra Madre Oriental and the Sierra Madre del Sur. The Sierra Madre Oriental presents a significant barrier to communications between Oaxaca City and the national capital, Mexico City, which lies 450 km, and five hours by road, to the northwest.

2. The Pacific coastal municipality of Huatulco which forms part of the narrow coastal lowlands between the Pacific Ocean and the Sierra Madre del Sur. The Sierra Madre del Sur presents a significant barrier to communications between Huatulco and Oaxaca City, which lies 250 km and six hours by road to the northwest.

As a consequence of these three sierras, Oaxaca's terrain is highly accidented and with altitude varying from sea level, along the Pacific coast, to 3750 m.a.s.l. at the peak of Mt Quiexobra in the Sierra Madre del Sur. Over 60% of the state is above 500 m.a.s.l. The varied orogeny of Oaxaca and the associated variation in rainfall, temperature and soils are the principal determinants of the high biodiversity of the state. Vegetation types range from lowland seasonal broad leaf forest in the Pacific lowlands to lowland tropical rainforest on the Gulf coastal plain adjacent to the state of Veracruz, and from pine and oak dominated forest in the mountains to cactus dominated matorral on the dry slopes of the Sierra Madre Occidental. Oaxaca is considered the most biologically diverse of the 38 Mexican states, a country which itself is counted amongst the five most diverse in the world (Mittermeier & Mittermeier 1992).

Climate

Oaxaca's climate is affected by the weather systems of both the Gulf of Mexico and the Pacific. For approximately half of the year between November and May, systems emerging from the Gulf predominate. However, rain coming from the east falls predominantly in the sierras leaving a rain shadow across the majority of the south and west of the state. Conversely these areas receive substantial rainfall for the rest of the year, and occasional hurricanes, from the Pacific. The result is that a seasonal climate predominates in most of the state with up to 90% of rain falling between the months of May and November (Álvarez 1998). The exception is the extreme northeast of the state where year round rainfall arrives from the Gulf of Mexico.

On the Pacific coast this seasonal climate regime is the principal factor determining vegetation type. The forests of the low-lying areas (< 350 m.a.s.l.) of Huatulco are 'seasonally dry tropical forests' (Mooney *et al.* 1995) here referred to as tropical dry forest. This forest type is the principal biological focus of this thesis. Annual rainfall distribution for Puerto Angel (40 km west of Huatulco,) and Pochutla (10 km north of Puerto Angel) is shown in figure 3.3, and corresponding variation in mean monthly temperatures is shown in figure 3.4. Mean annual precipitation is markedly variable: for the period 1989 to 1996 for Puerto Angel it was 1127 mm but varied between 734 mm and 1 542 mm in the driest and wettest years respectively (INEGI 2001). The effect of this seasonality on the forests of Huatulco is that growth is limited to the wet season and the majority of trees are deciduous, losing their leaves at the beginning of the dry season and not flushing new ones until just before the rainy season commences.

Figure 3.3 Mean monthly precipitation for coastal Oaxaca. Source: INEGI (2001)

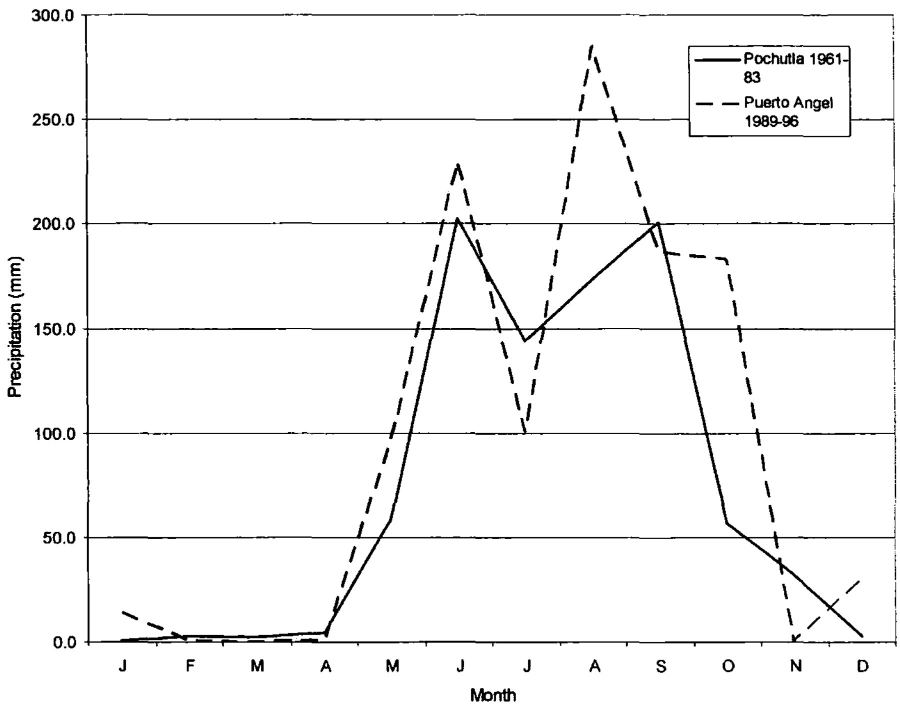
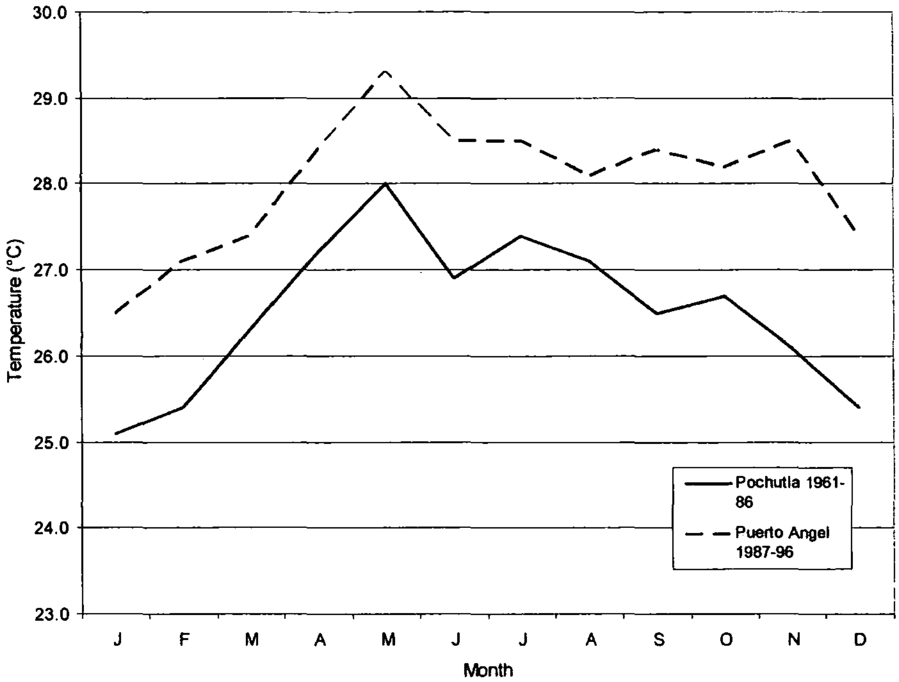


Figure 3.4 Mean monthly temperatures for coastal Oaxaca. Source: INEGI (2001)



3.3 Economic Geography

For most of its colonial and postcolonial history, Oaxaca has been economically marginalised from Mexico City primarily because of a lack of natural resources suitable for rapid exploitation. Two relatively brief periods were exceptions to this. In the early colonial period, Mexico's major port on the southern Pacific coast - and therefore the main connection to Central and South America - was Santa Cruz de Huatulco. This, and the associated trade route which went through Oaxaca City, brought considerable importance, and some wealth, to the state. However, the establishment of better communications with the Pacific port of Acapulco in neighbouring Guerrero meant that by the end of the sixteenth century this latter port had taken prominence. In the mid-eighteenth century the production of cochineal (a red dye made from insects grown on native *Opuntia* cacti) grew rapidly in response to demand from the textile producers of Europe (Murphy & Stepick 1991). This stimulated economic growth, particularly in the *Valles Centrales* around Oaxaca City. However, the War of Independence (1810-1821) interrupted production and export, with much trade being lost to Guatemala which was to become the world's major exporter of this dye until the introduction of synthetic substitutes in the middle of the 19th century.

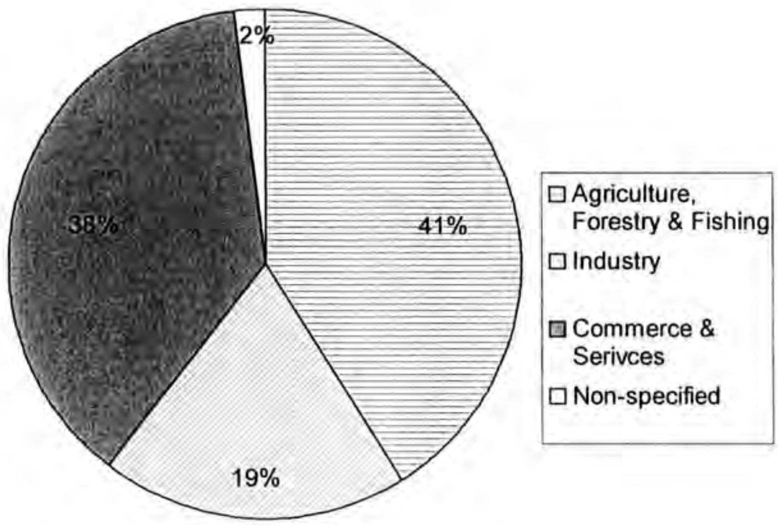
Industrial investment in the state came briefly to Oaxaca in the early twentieth century in the form of the construction of the trans-isthmus rail link from the Gulf Coast of Veracruz to the Oaxacan port of Salina Cruz. However, construction was disrupted by the Mexican Revolution (1910-1917) and the link was superseded by completion of the Panama Canal. Today Salina Cruz is a petroleum processing centre and therefore one of Mexico's most important ports as well as being Oaxaca's only example of large-scale industrialization. Crude oil is pumped directly form the Gulf Coast across the Isthmus of Tehuantepec to Salina Cruz and output from the refineries is sent by tanker to markets on the Pacific coast of the Americas. Today only 19% of the economically active population of the state work in industry, agriculture and commerce being the main employers, see figure 3.5.

Outside of the *Valles Centrales*, agricultural production has been, and remains, only partially capitalised, and in much of the highlands, where indigenous communities dominate, peasant modes of production continue. In many communities this is now supplemented by transfers of earnings from community members working in Mexico's urban centres and the USA.

Agriculture in the state remains dominated by production of Mexico's staple, maize. In 2000, 49% of Oaxaca's cultivated land was under this crop, with coffee (15%) and beans (4%) following in importance (INEGI 2001). Coffee was introduced to the southern slopes of the Sierra Madre del Sur in 1885 and it is still an important export crop. However, coffee producers remain largely isolated economically from the state capital with coffee leaving Oaxaca by the southern ports and overland to Veracruz on the Caribbean coast.

In Huatulco the area under maize is in decline. During the period from 1996 to 2000 it fell from 46% to 38% of cultivated land, and is now superseded by the area under coffee (51%) the absolute area of which has remained relatively stable (INEGI 2001). A similar fall in maize cultivation is not detected at the state level. The change in Huatulco probably reflects the combined effect of the decline in government price support for maize and the financially more attractive employment options offered by Huatulco's growing tourist sector. It should be noted that coffee production, unlike maize, is dominated by relatively few large, private farms and as a perennial crop with higher establishment costs is perhaps less likely to be rapidly taken out of production.

Figure 3.5 Waged employment by sector in Oaxaca. Source: INEGI (2001)



In concluding, Clarke (2000 p. 249) suggests that:

> Weak integration of core and periphery on the Mexican national scale has left Oaxaca in an incipient stage of capitalism.

Oaxaca therefore remains 'on the peasant periphery' of Mexico. However, Huatulco appears relatively prosperous in comparison with the rest of the state. In 2000, 15.8% of salaried workers in Huatulco earned below the minimum daily salary of US$ 3.44, compared to 47.9% in the state of Oaxaca as a whole (INEGI 2001).

3.4 Social and Political Geography

3.4.1 Oaxaca City and the State of Oaxaca

Oaxaca is ethno-linguistically the most diverse Mexican state with sixteen language groups of prehispanic origin, many of which are subdivided into mutually unintelligible dialects, still spoken in the state. Some 40% of Oaxacans speak an indigenous language, compared to a national average of 10% (Clarke 2000).

Spanish is, however, the *lingua franca* and dominates in Oaxaca City and other larger towns. The ability to speak a native language is the most basic marker of 'indigenousness' in a mestizo dominated state and nation, and the comparatively high numbers of speakers of indigenous languages is a reflection of the degree of Oaxaca's history of economic marginalisation.

Economic separation from the rest of Mexico has also been reflected in political separation. This is demonstrated by Oaxaca's peripheral involvement in the Mexican Revolution. Whilst Oaxaca saw continuous minor skirmishes between supporters of the revolutionary factions, there was no mass of landless peasantry, as there was elsewhere, to lend both motivation and support to the revolutionary movement of Zapata (McLynn 2000). None of the major battles of the revolution were fought in Oaxaca. The most significant development of this period being the creation of a conservative led Oaxacan Sovereignty Movement that claimed, in 1916, to have ceded from the rest of Mexico in response to the dictatorship of Carranza. Carranza thought this of so little import that he chose to ignore it for several months before his forces rode into Oaxaca and quickly defeated the secessionists. As Murphy and Stepick (1991 p. 43) conclude, Oaxaca remained distinct from the rest of post-revolutionary Mexico in having:

> [N]o large landed class, no large numbers of dispossessed peasants, no revolutionary heroes, no beneficiaries of the revolution, and no seeds for industrialization.

The federal government's influence over Oaxaca City has grown during the latter part of the 20th century as the dominant political party, the Partido de Revolution Institucional (PRI) has pursued its twin aims of urban industrialisation and reconstruction of the peasantry through land redistribution. The result has been increased urban-rural differentiation, driven not by industrialisation but by 'bureaucratization' associated with the increasing reach of state services and the growth of commerce in Oaxaca City and other major urban areas (Clarke 2000).

3.4.2 Huatulco and the Pacific Coast

Whilst Oaxaca City and the surrounding *Valles Centrales* have historically had poor political and social ties with Mexico City, the Pacific coast of Oaxaca, has been further marginalized from the *Valles Centrales* due not least to the physical barrier of the Sierra Madre del Sur.

Industrial development in Salina Cruz has had little effect in Huatulco 200 km further west along the coast. For much of the twentieth century Huatulco was relatively under populated with dispersed agrarian communities. These communities typically managed land under communal forms of tenure or worked on the large estates of local *caciques* (politically strong rural 'chieftains').

The area of communally controlled land on the coast increased during the presidency of Lazaro Cardenas (1934-40). During this time, the promise of land reform that had been one of the major outcomes of the Revolution began to be partially fulfilled and *ejido* communities (communities in which land is controlled by the community under a system of non-transferable title) were established along the coast. However, already existing forms of communal tenure in Huatulco continue to predominate there.

The various hamlets in Santa María Huatulco were established largely by within-state migration of *mestizos* (people of mixed Spanish and indigenous origins) from the district of Miahuatlán, at the southern end of the *Valles Centrales* where a shortage of water has always made agriculture a marginal activity. The main influx of population from Miahuatlán occurred from the mid 1940s to mid 1960s. Migrants typically moved south, first to work in the coffee farms of the Sierra Madre del Sur and then, hearing of the availability of land on the coast, moved down to settle there.

The 'opening up' of the coastal region began in the early 1980s with the completion of improvements to the Pacific highway that now connects the entire coast of Oaxaca with the rest of Mexico's Pacific states. Later in the decade construction of the *Bahías de Huatulco* tourist complex began, a development aimed at capturing out-of-state tourists, both national and international. Federal support has encouraged large investments from international hotel chains along the picturesque coastline, and smaller scale investment in cheaper hotels and restaurants in and around the service town of La Crucecita, principally by immigrants from elsewhere in the state and country. The influx of people into the municipality to pursue work and investment opportunities has resulted in population changes in Santa María Huatulco that are quite different to that typical of the rest of the state (figure 3.6).

Figure 3.6 Population trends in Santa María Huatulco and Oaxaca State 1950-2000. Source: INEGI (2001)

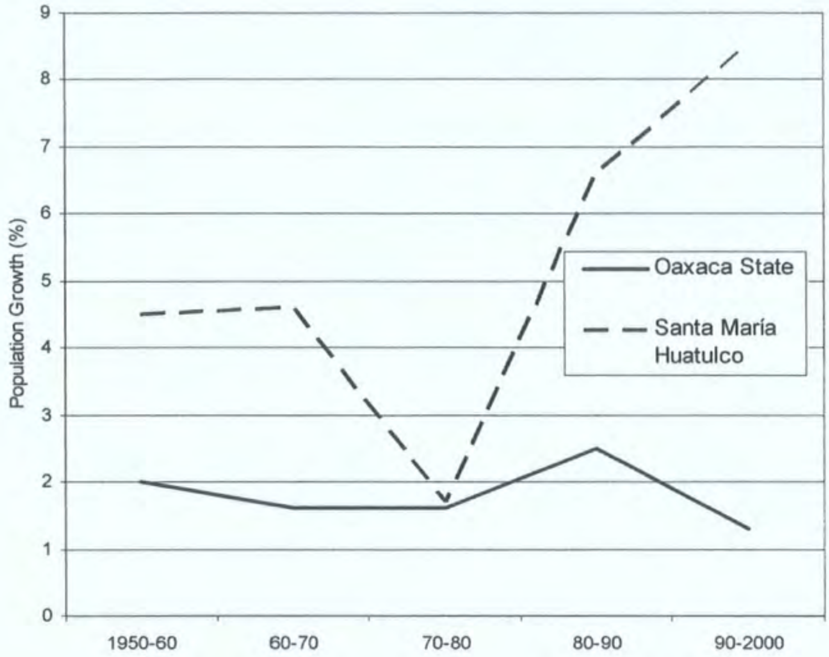
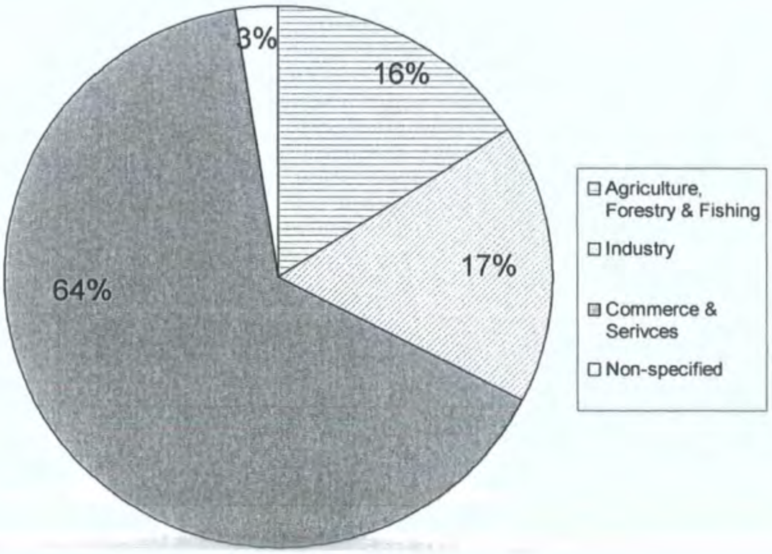


Figure 3.7 . Waged employment by sector in the municipality of Santa María Huatulco. Source: INEGI (2001)



Several established agrarian communities had their land expropriated for this development and have been relocated further inland to areas of mature seasonally dry tropical forest. In these new locations existing communal management structures have been reconstituted. Communities that neighbour the tourist development have not had access to sufficient capital to become investors themselves, but play an important role as a source of cheap labour in the construction and service sectors. For many members of these communities employment related to tourism is now a highly attractive option. The waged workforce has undergone dramatic restructuring, with agriculture now a minority pursuit in the municipality (see figure 3.7). However, traditional agrarian management structures and rights to communal land have not been given up; the security offered by farming has not lost its attraction to Huatulco's *campesino* communities. The diversity of the dry forest landscape of Huatulco and coastal Oaxaca is illustrated in figures 3.8 to 3.13.

3.5 Discussion

Murphy and Stepick (1991) identify a recent trend towards successful imposition of the Mexican state on Oaxacan affairs. Attempts at such imposition, whilst common since the time of Cortéz, had previously been half-hearted, and hence easily resisted, partly because of Oaxaca's isolation and partly because of the relative unattractiveness of its largely rural economy (compared to urban-industrial centres such as Monterrey, Guadalajara and Toluca). Integration into the broader national economy is now continuing apace and is exemplified by recent investments in transport infrastructure. Two international airports have been built, first in Oaxaca City and more recently in Huatulco, and currently the Pan-American Highway is being upgraded between Oaxaca City and the Isthmus of Tehuantepec. This latter project is part of a drive to integrate the economies of Southern Mexico and Central America under the 'Plan Puebla-Panama'.

This current trend contrasts with the same authors' conclusions concerning Oaxaca's history. They argue, in broad agreement with Clarke (2000), that this history has been one of marginalization from the 'core' of Mexico City since colonial times, and probably before that. The constant during these times has been Oaxaca City's central role as a marketing and administrative centre for the agrarian communities that continue to make up much of the rest of the state. Imposed on this constant have been cycles of increasing and decreasing economic links with the rest of Mexico as outside elites have exploited the surpluses derived from temporary expansions of Oaxaca's economy. Previous examples of this already noted include Oaxaca's importance as a trade route to the Pacific and the cochineal trade. To these can be added sporadic booms in mining activities and the current growth in tourism in Oaxaca City, the *Valles Centrales* and along the coast.

The current 'boom' in interest in Oaxaca's biodiversity has followed a similar pattern- it is driven by outside elites some of whom seek the employment opportunities offered by increased international and national funding for biodiversity conservation. In turn these elites have brought to bear on Oaxaca's forests national and international nature conservation norms. In Huatulco this interest in biodiversity conservation is made more urgent by the rapid changes provoked by tourist development, indeed it can be argued that Oaxaca's dry forests are entering a period of uncertainty unlike any other in their history, one in which international tourism, international conservation politics and local land management practices jostle for influence. The perceived need to quantify and communicate the value of dry forest diversity in Oaxaca to global stakeholders is the stimulus behind the biodiversity assessment that is the subject of chapters 4, 5, 6 and 9 of this thesis. The resulting social relations are the subject of chapters 4, 7 and 8.

Figure 3.8 Tropical dry forest in Huatulco at end of dry season. Note evergreen riverine forest.

Figure 3.9 Forest fallow, Huatulco. White flowered tree is the pioneer species *Cordia elaeagnoides*

Figure 3.10 Recently established dwelling in dry forest of Huatulco.

Figure 3.11 Abandoned dwelling and regenerating dry forest, Huatulco.

Figure 3.12 Dry forest on rocky outcrops on coastline of Huatulco

Figure 3.13 Tourist development in dry forest zone, Huatulco

Chapter 4: Performing Biodiversity Assessment

4.1 Introduction

That biodiversity is the dominant conceptualisation of nature amongst biodiversity scientists is now indisputable. As Callicot (1999 p. 23) put it, it is the '*summun bonum* of conservation biology'. Biodiversity assessment is the practice of comparing the biota of one location with that of another, or others, and is used to inform decisions on what to conserve, where to conserve it and when. It is, hence, a necessary part of modern nature conservation. It is also a subject worthy of geographical analysis in that it summons the ontological and epistemological resources of modern biology to describe new territorialities (e.g. ICBP 1992; Myers *et al.* 2000). Given the magnitude of the threats to biodiversity that have been identified by scientists (Janzen 1986; Myers 1979; Nabhan & Antoine 1993), and the inevitable scarcity of resources to confront those threats, appropriate conservation strategies must be based on informed, rational decision making. Hence the theoretical importance of assessments, even if in practice they are not always carried out (Dudley & Jeanrenaud 1998; Vanclay 1998). Assessment may be done at any level of biodiversity, that is genetic (e.g. Blaxter & Floyd 2003), species (e.g. Gordon *et al.* 2004) or ecosystem (e.g. Dinerstein *et al.* 1995). However, as Gaston (1996a) argues, it is the species that is most often regarded as the fundamental unit of biodiversity [see also Colwell and Coddington (1994)], estimates of numbers of species are the most widely applied and understood measures of biodiversity and species loss the main manifestation of the biodiversity crisis. Assessment may be done directly by locating, identifying and enumerating species or indirectly using surrogates that may vary from an individual indicator species to patterns of remotely sensed radiation. Given the resource demands inherent in surveying tropical habitats and identifying large numbers of species there is much interest in the use of surrogates of species diversity and their potential to reduce the time and cost of performing assessments. However, as yet no entirely satisfactory surrogates have been found (Armstrong 2002; Caro & O'Doherty 1998) and direct measurement of species diversity remains the most widely applicable approach to species diversity assessment. Hence, it is the direct measurement of species diversity that is the subject here and, as an inevitable consequence of disciplinary specialization, tree diversity in particular.

The above qualifications are in themselves indicative of the difficulties inherent in assessing biodiversity. No tropical biologist is capable of identifying all the components of nature (e.g. neither plants and insects, nor genes, species and ecosystems) and no assessment technique is suitable to capture all these diverse natural forms. The ways in which biodiversity is sampled and the types of comparisons that are made between species and assemblages of species are as much defined by the interests, abilities and politics of the assessors as they are by any innate characteristics of the biota in question. Yet defining the 'what' and the 'how' of biodiversity assessment are practical and theoretical necessities, hence Sarkar's (2002) contention that biodiversity is not pre-defined, but defined by how we choose to measure it.

In this scientific pursuit positionality determines the models proposed for testing (Hayles 1995) and ontological and epistemological compromises have to be made from the outset, raising the question of how biodiversity assessments can lay claim to authority. This chapter therefore explores biodiversity assessment and the processes by which it becomes authoritative. It is based on participant-observation of a tree diversity assessment carried out in the tropical dry forests of Oaxaca, southern Mexico. Its aim is to elucidate the extent to which the concept of biodiversity is meaningfully reproduced amongst actors in southern Mexico. This is done by consideration of some of the networks of social relations within which *Naturaleza*[5], a Mexican non-government organisation (NGO), and biodiversity scientists operate. By juxtaposition of this scientific representation of nature as biodiversity with more locally situated ones consideration is given to how 'facts' in conservation biology come into being and circulate along side, or at the expense of, local ones. This chapter shows that the assessment of biodiversity, whilst being site specific, is conceptually very distant from the understandings of humans living with/managing the biodiversity in question. In conclusion the politics of biodiversity assessment are discussed from a postcolonial perspective.

[5] Pseudonyms are used for this institution and its staff

4.2 Background to an Assessment

4.2.1 People and Places

Between 2002 and 2003, a team of three people carried out a biodiversity assessment of the trees and shrubs of the tropical dry forests of the municipality of Santa María Huatulco in Oaxaca as part of an exercise to evaluate current conservation efforts there. The team consisted of me, a 'northern biodiversity scientist', and two staff from *Naturaleza*, Pedro and Jorge. Pedro and Jorge, residents of the small market town of Santa María, were born in the vicinity into resource-poor farmer (*campesino*) families and both continued to meet part of their household needs through farming. However, by the time this assessment took place, an important part of their incomes was earned working as technical staff (*tecnicos*) for *Naturaleza*. *Naturaleza* supports locally led conservation and rural development initiatives in the dry forest areas of coastal Oaxaca, principally in Santa Maria Huatulco. Its head office is in the state capital, Oaxaca City, 250 km north of Huatulco where the majority of the senior staff, all university graduates, are based. Despite growing interest in the diversity of this region and several years of financial support from national and international conservation interests, *Naturaleza* had not previously carried out a 'scientific' assessment of the biodiversity of Huatulco.

Pedro and Jorge were mestizos, i.e. of mixed Spanish and Amerindian stock. Along the coast of Oaxaca there are several ethno-linguistic groups represented, but my co-workers' origins in these groups were now unclear and they spoke only Spanish. Both had also lived and worked outside of Oaxaca, Pedro briefly in Mexico City and Jorge illegally in the USA.

Within *Naturaleza*, Pedro and Jorge were amongst the most long-standing of the technical staff, all of whom are locally recruited. Jorge was the curator of the small but growing 'Community Herbarium' and Pedro was in charge of botanical collecting. In practice, their work was interchangeable; they usually went together to the field on collecting trips and worked together on processing (drying and cataloguing) botanical specimens.

4.2.2 Tree Spotting

When assessing tree diversity in the forest, Pedro was indispensable because of his ability to distinguish between trees and name them, by their local names, consistently. In this part of Oaxaca previous assessment exercises have identified in excess of 200 species of trees and shrubs (Gordon *et al.* 2004) and assessment is entirely dependent on identifying species. In a diverse tropical forest where at any given time many of the trees can be without fruit, flower or leaf, this is a considerable skill. From the perspective of the university trained biologist, the distinction between species lies in the precisely defined characters of the reproductive organs, primarily the flower but also the fruit. However, each species has a set of *field characters*, traits that are easily seen in the forest and useable whether a tree is in flower or in fruit or not. These traits include leaf characters, bark texture, the angle of branching and the smell or colour of the underbark. Pedro was a master of recognising them and, of equal importance, of recognising them quickly. He was capable of considering many of these characters at once and hence recognising a hard to define *essence* of each species. This he presumably learnt from a long and close association with the forests of the area. His ability to do this certainly pre-dated his work with visiting scientists.

What is important to note here is that there was a departure between what Pedro, as a local farmer, and I, as a biologist, regarded and used as knowledge about the identities of trees in Huatulco's forests. Where this departure leads will be further considered, for now it is important to emphasise that Pedro knew trees in a way that was very different to the way that I had come to know them.

4.2.3 Sampling Tree Diversity in a Mexican Forest

A typical scene from a forest on a hot afternoon would have Jorge cutting a 50 metre transect through the undergrowth to form the line along which a plot was to be measured. Pedro walked that line calling out the local names of those trees whose trunks fall within a set distance from the transect line. In this way a defined area, a 100 m^2 plot, was demarcated and enumerated. Plots are useful tools in plant diversity assessment as they control sampling effort, ensuring that similar amounts of effort are expended in each forest and hence that results are comparable. I would follow Pedro, glancing at each tree to check that the name he had given me appeared to have been used consistently for the species in question. Occasionally I challenged him, usually he proved to have been consistent. I would then record the presence of the tree on a field sheet, usually using an abbreviation of its scientific name. Sometimes Pedro would not recognise a tree as one for which he had a name, so we would pause while he removed a branch, preferably one with flowers and/or fruits, to make a botanical collection for later identification by experts in Mexico City. Thus, based in no small part on the abilities of Pedro, a set of 'scientific facts' would begin to emerge from the forest. At that moment, between Pedro calling out a local name and me inscribing its presence on a fieldsheet, a transformation took place that spoke of, amongst other things, scientific privilege and power, history, postcolonialism, the production of knowledge, the construction of nature and the mobilisation of networks. All this occurred in an instant, in a patch of unremarkable looking forest, in an obscure corner of a poor state in southern Mexico.

4.3 Tree Names and the Networks they Inhabit

4.3.1 The Transition from Local Names to Scientific Names

The scientific name of an organism, usually presented in binomial form, is familiar to any good student of biology in 'the North' before she or he leaves school. Each animal or plant that has been subject to investigation by a taxonomic scientist will have a scientific binomial of the sort formulated by Carlos Linnaeus, the 18th century Swedish biologist. The binomial is made up of the genus and the species and it is unique, distinguishing a species from every other on the planet. When Pedro called *guanacastle* I would know from previous experience that we were confronted with *Enterolobium cyclocarpum* (Jacq.) Griseb, a 'leguminous' species of the family Mimosaceae, whose original description was by Nicolaus von Jacquin with a later repositioning of the species by August Grisebach. What is interesting is why it is that I should need, as a scientist, to translate from *guanacastle* to *Enterolobium cyclocarpum*.

Local names (or 'folk-taxonomies') correspond to Foucault's 'classical episteme' in that:

> [T]he sign can be more or less probable, more or less distant from what it signifies, it can be either natural or arbitrary, without its nature or value as a sign being affected- all this shows clearly enough that the relation of the sign to its content is not guaranteed by the order of things in themselves. (Foucault 2002 p. 70).

Thus *Swietenia humilis* Zucc., better known to the English speaking world as mahogany and to much of the Spanish speaking world as *caoba*, is known in this part of Oaxaca as *Palo de Zopilote*: vulture tree. The large smooth seedpods hang heavily from up-turned branches in a fashion that recalls the drooping head of a vulture.

Foucault shows that the transition from ordering objects by 'similitude' to one of ordering objects by how they are distinct from other things is a change of difference and identity. He argues that this process, which he traces back through Bacon and Descartes, is one that marks the beginning of the modern world. Scientific names are to him undeniably a product of a mode of thinking that emerged in Western Europe. *Enterolobium cyclocarpum* does more than just signify a particular species of tree, it also tells us about its relationship with other more or less similar species. I know from its name that it is a 'leguminous' species and thus shares certain 'fundamental' or homologous characters with, for example, pea plants but not with mahogany. The fact that *guanacastle* and *mahogany* share the very obvious but analogous characteristic of being trees, (unlike pea plants) is irrelevant to this classification. Foucault (2002 p. 74) concludes that in the modern world 'similitude', it is now a spent force, outside the realm of knowledge. It is merely empiricism in its most unrefined form.

Should I wish to use the name *guanacastle* in a scientific publication it would be considered inadmissible, non-knowledge, not withstanding that it represents Pedro's considerable skills and knowledge. The acceptance of Linnaean binomials in scientific discourse has become universal, and in turn it is their universal acceptance that is considered to be their advantage over local names. In the biodiversity assessment I was re-enacting in a geographical context (periphery to core) the same transition that Foucault describes in an historical context (classic to modern). Any vestiges of similitude in the Linnaean binomial are redundant; the biodiversity scientist can complete his work ignorant of any reference to a plant's appearance that may be hidden in the binomial. (The specific epithet *cyclocarpum* refers to the distinctive curled legume of guanacastle.)

4.3.2 Why Scientists Use Scientific Names

I needed to foster this transition in order to give these forests a voice in the networks that promote biodiversity conservation. My concern was to produce a piece of science that my peers and colleagues would respect, not one that Pedro or Jorge would understand or recognise. To do that I had to, in the language of Latour

(1987), create a representation, or representations, of the forest that surrounded us. Such representations needed to be robust enough to leave this peripheral place and enter into a transnational network whose gate keepers are the editorial boards of scientific journals. Once in that network they might influence a community composed of other scientists, transnational non-governmental organisations (e.g. WWF, Conservation International, The Nature Conservancy) and government environmental advisors. Together these institutions and individuals form an epistemic community promoting a global agenda for the conservation of biodiversity (Haas 1992; Raustiala 1997a).

Given that these representations attempted to describe forests as biodiversity they can be said to be performative in that the practice of biodiversity assessment brings the biodiversity of the forest into being. Biodiversity is a scientific concept, requiring scientific description and Law (undated) argues that every description is performative, tending to bring into being what is being described. Within the networks of the epistemic community of the biodiversity agenda the representations would have to continue performing; they would be interpreted and reinterpreted, compared and contrasted, used partially or entirely, in an 'incessant presentation' (Dewsbury *et al.* 2002 p. 438). Should a representation succeed in these continuous performances the cause of biodiversity conservation might be furthered. It is this ethical commitment that defines the epistemic community and, by practicing a biodiversity assessment, I become part of that community. The translation from the local to the scientific name is the first stage step in this long process.

The Linnaean system of biological nomenclature is the gateway to describing and quantifying nature as biodiversity and therefore the way in which scientific discourses of nature are framed. Biodiversity scientists would argue that the binomial system has three great advantages over local names or 'folk taxonomies'. First, they remain stable across linguistic divides, unlike *mahogany/caoba/palo de zopilote*. Secondly, they convey more information about the relationship of a species with respect to its relatives; hence we could expect that *Swietenia humilis* would have much in common with its geographically distant relatives *Swietenia macrophylla* and *Swietenia mahogoni*, and are therefore not surprised to learn that their timbers are similar. In other words, and to paraphrase Foucault, the relation of the sign to its content *is* guaranteed by the order of things in themselves. Thirdly, we can test our belief/establish the fact that the tree in front of us is what we think it is by reference to a 'type' specimen stored in distant herbarium, or to the formal description of that type, which in the case of *Swietenia humilis* was done by Joseph Zuccarini. A complex set of social and natural articulations has brought this fact into existence with the result that the correspondence between nature and language *appears* to be achieved by a single leap- as soon the tree is given a name, and not before, its existence becomes a scientific fact. This, Latour (1999) argues, is indicative of the old 'modernist settlement'. The set of articulations that make that leap possible are hidden, or forgotten, with the result that the assumed realism or objectivity of that correspondence goes unquestioned.

The reasons for the preference for the Linnaean binomial were not of immediate relevance for me as we assessed forests in Oaxaca. What was relevant to me was the power of the scientific name over the local one. The Linnaean binomial allows the 'fact' of this species' presence in Oaxaca to be mobilised but at the cost of that fact becoming a privileged and exclusive form of knowledge. There is a postcolonial aspect to the production of such scientific knowledge in that it is considered rational and powerful in contrast to the 'other' subaltern knowledge that is indigenous to the location of the forest (Blunt & Wills 2000). This of course is not a phenomenon unique to biodiversity assessment, or even science in general.

4.4 Mobilisation and Circulation

4.4.1 The Legitimisation of the Botanical Fact

Before we could truly mobilise our scientific fact- that a certain tree exists in a certain forest- there was more work to be done. We needed to legitimise that fact by the creation of an artefact. *Amphipterygium adstringens* (Schlecht.) Schiede, is a characteristic tree of the tropical dry forests of Mexico's Pacific lowlands having a natural distribution from the north-west of Guatemala to the north-west of Mexico. Like most of the trees found in these forests it is drought deciduous losing its leaves for most of the six month dry season. It has distinctive short thick branches and a spiny reddish brown bark that has a bitter taste. It is usually placed by taxonomists in the family Julianaceae, a monogeneric family, but it is has previously been included as a member of the much larger Anacardiaceae, the family of mangos, sumacs and cashews.

During work done in Oaxaca before the assessments carried out with Pedro and Jorge, I made a 'collection' of this species in the company of Ricardo, a botanist from the National Herbarium in Mexico City (MEXU by its standard abbreviation). Ricardo, a Spanish speaker, was born and raised in Mexico City and had worked for many years in this herbarium which is located within Mexico's largest and most prestigious university. MEXU holds one of the world's largest collections of dead plant parts, housing nearly a million specimens. This herbarium is organised and run just as are the great herbaria of Europe. As an institution it is similar to a particular type of European institution, the great herbaria, that rose to prominence in the 18th and 19th centuries.

We made a specimen of *Amphipterygium adstringens* by taking a leafy branch with fruits, putting it in a press and giving it a unique identifying number. The aim of this, which was done whilst still in the forest, was to enable the tree's confident identification with respect to its Linnaean binomial. The specimen became our representation of a species that Pedro knows as *cuachalalate*. The task could only be formerly accomplished when the specimen arrived at MEXU. Our artifactual representation of this tree could not leave the work benches of the herbarium, and enter the collections proper, until it had been assigned, correctly or otherwise, a binomial- it had to continue to perform the translation from *cuachalalate* to *Amphipterygium adstringens* on the workbenches of the herbarium. Specimens in MEXU are arranged with respect to the hierarchical system founded, although since much changed, by Linneaus. Without the binomial we simply would not know where to put it, with the binomial we could place it in the correct folder on the correct shelf in the correct room in the herbarium. The binomial was, in effect, the gatekeeper to the herbarium collection. The specimen, mounted on cardboard, now carries its name proudly at the top of its label, with mine and Ricardo's names as collectors and Ricardo's as the person who provided the formal 'determination' of the Linnaean binomial. Once this had been done the transformation of the tree was complete; it had become an artefact and could now be stored in the correct place in the herbarium. It now sits in the herbarium accessible to any botanist who cares to visit, and by the system of inter-herbarium loan, to any botanist in any other herbarium. From that original collection we made in the forest we had enough plant material to make a duplicate, to which we attached a copy of the label, and sent it to a collaborating herbarium in Honduras. In this way we established the botanical 'fact' that *Amphipterygium adstringens* existed at a certain time in a certain place. Our botanical collection took on all the characteristics of an 'immutable mobile'- held together in a network of articulations, remaining unchanged as it be moved from place to place, transporting a representation of a distant phenomena (Golinski 1998;

Law 2000). The botanical collection speaks to the world of the fact that *A. adstringens* existed in a certain place at a certain time. Should anyone question our fact, we would direct the interrogator to this collection, rather than the trees in the forest from which it came, some 1000 km away.

This process required the mobilisation of many resources including that of two herbaria, the authority of Ricardo as a respected botanist, the original work of Schlechtendal and Schiede, the funders of this particular research (The British Government), the research institute to which I belonged, and a Western intellectual tradition that stretches back centuries. We did not so much discover a fact as manufacture it by the production of an artefact through processes of transformation and mobilization (Law 1994). This was the result of a highly socialized process, and it is in this socialization that its strength lies. As Latour (1987 p. 61) points out, it is this ability to mobilize resources 'on one spot' that makes scientific rhetoric particularly powerful.

However, we mobilized very little that was Oaxacan in the creation of the representation of a Oaxacan tree. The resources we mobilised led to the accumulation of facts, in the first instance, in far away Mexico City where only a distant echo remained of Oaxaca in the form of a note of the tree's local name, *cuachalalate* on the specimen label. Most collections do not carry a note of the local name; it is not considered essential information.

We re-established the 'fact' of *A. adstringens*' existence in the forests of Santa Maria Huatulco, whilst carrying out the biodiversity surveys with *Naturaleza*. Whilst its representation as a botanical specimen in the herbarium may remain fixed, for decades or even centuries, our fact underwent another translation, this time from field sheet to computer. The computer then became, in Latour's (1987) terms, a producer of inscriptions, and it is inscriptions 'that provide scientists with their final source of strength' Latour (ibid p. 90). I compared lists of different species from different forests by various forms of analysis and in so doing produced the graphs and diagrams that become representations of forests (see chapter 9). Then I was emboldened to take the highly political step of weighting each species according to how important each was as an object for conservation (rare species are more important- they need more of somebody's resources if they are to be conserved). Graphs, scatter plots and weighting allow me to propose, through publications and conferences, that some forests contain more important biodiversity than others (e.g.

Gordon *et al.* 2004), and in an extension of the highly political business of weighting species I was able to weight forests, that is, I proposed which of the forests ought to be conserved. Biodiversity assessment therefore has a very literal parallel with Foucault's project to show that the 'organization of species maps directly onto the organization of space' (Elden 2001 p. 127) and it is in this sense that biodiversity assessment defines space (Massey 1991) and converts it to place. These new places are unrelated to previous territorialities but become politically defined as priority ecosystems and hotspots (e.g. Dinerstein *et al.* 1995; Myers *et al.* 2000) in need of conservation. Pedro, Jorge and their fellow residents of Santa María Huatulco, to whom these forests belong, were now forgotten. *A. adstringens* as an actor in the process continued on its way, along with many other species, buried deep in various inscriptions. Its presence was still felt in the distribution of dots on a scatter plot or data in a table (see chapter 5 for examples). Such inscriptions are the lifeblood of publications on the conservation of biodiversity and typically serve to summarise and simplify the complex data sets that result from sampling diverse tropical forests. They have authority as manifestations of biodiversity science and can circulate in the politically connected epistemic communities (Nelson 1997; Takacs 1996). The inscriptions produced had not just left the residents of Huatulco behind, also left were *cuachalalate*, *guanacastle*, *palo de zopilote* and all the other local names of trees along with their uses and histories. This barrier to understanding is rarely acknowledged, yet it is of profound importance and goes to the heart of the practice of biodiversity assessment. Conservation scientists are political actors, influencing resource allocation decisions remotely in ways that the residents of Santa María Huatulco cannot because the inscriptions that inform the debate are unintelligible to them and effectively beyond their reach. They are excluded from the decision making processes that can result in such profound changes to their environment as the declaration of the Huatulco National Park within their municipality which has effectively barred them from 6 000 ha of land to which they once had access.

However, a moment's reflection shows just how culturally situated is this scientific approach to nature's diversity. Consider the binomial *Amphipterygium adstringens*. Its etymology is Greek, meaning two winged seed and *adstringens* refers to its bitter bark, (about which more below). Greece was the founding culture of modern western biology and taxonomy (Mayr 1982); the very notion that a species might be a discreet and delimited entity has its origins in Platonic and Aristotelian essentialism. The original description of the species was written in Latin. Linnaeus, a

Swede, popularised a hierarchical and relational taxonomy. Darwin, an Englishman, gave to the Linnaean pattern a process called evolution. The point, as Escobar (1999) argues, is that nature is constructed, or represented, according to the position of the observer with the result that most social groupings, including biodiversity scientists, inevitably have a view of nature that is a hybrid of cultural and natural phenomena. This hybridity is inevitably reflected in the nomenclatures that describe nature. Biodiversity scientists' representations are powerful and useful but they are at the same time hybrid, and no more real and no less constructed than those of any other social group. Their power comes from their ability to circulate through privileged centres of calculation and epistemic communities.

4.4.2 Mobilising the Local Name?

Juxtaposed against the Euro-centrism encoded in of *Amphipterygium adstringens* is *cuachalalate* whose etymology lies somewhere in the tangled mass of Mexico's pre-Colombian languages. Santa María Huatulco was alternately predominantly Zapotec and Mixtec in ethno-linguistic origin but the natural range of the species stretches across the historical boundaries of many ethno-linguistic groups, across which the name *cuachalalate*, or variants of it, is well preserved. What can be teased out is Náhualt (Aztec) influence in its *–ate* ending, the same ending which reaches English in coyote and chocolate. The mobility of this name ending hints at a mobility that local names may have to rival that of their corresponding Linnaean binomials, whilst *cua*, or its variant *gua*, is common to many local tree names across Mesoamerica, (e.g. *guanacastle*) and reaches English in the tree product guacamole.

Cuachalalate has medicinal properties; an infusion prepared from its astringent bark is recognised as a cure for a variety of stomach ailments, including gastric ulcers. Its bark is traded in the distant markets of Oaxaca City and at greater distance still in Mexico City, the historic centre of Nahual culture, hundreds of kilometres from its native forests on the Pacific coast. It is always traded under one of the variants of its local name, as no doubt it has been since prehispanic times. On a different scale, and by a different medium, *cuachalalate* has circulated further still than the markets of Mexico City. A search conducted on google.com revealed 153 hits for *Amphipterygium* compared to 364 for *cuachalalate* [6]. This interest, which is derived largely from its medicinal properties, has allowed *cuachalalate* to circulate in international networks independently of, and more extensively than, its Linnaean

[6] Performed 29/01/2004

binomial. It is clear that non-scientific representations of trees are capable of circulating in networks both ancient and modern. The important distinction here is that the *cuachalalate* of the market places is undisguised in its hybridity. It is defined not in relation to similar trees but somewhere between its nature, (its 'treeness') and the cultures that recognise its medicinal properties.

4.5 Ontological and Epistemological Authority

The authority with which scientists speak is derived from their claim to objective rigour in the construction of facts about the world. However, the distinction between the efforts of the biodiversity scientist to represent nature and those of every other human is not as clear as is commonly supposed. The point was made earlier that the basic unit of biodiversity is the species. Yet the very concept of what a species is has taxed biologists from the earliest times to the present. As Mayr (1982 p. 251) notes 'There is probably no other concept in biology that has remained so consistently controversial as the species concept.' Yet, buoyed by calls to increase their efforts (e.g. Janzen 1994) taxonomists continue to name species, and biodiversity scientists continue to count them, as a practical necessity for the furthering of biodiversity science. Practicality demands that we do not await the resolution of a centuries old debate before the biodiversity crisis is tackled, so the assumption is made that each species encountered is a discreet entity. The same may be said of the other levels of biodiversity; in measuring ecosystem diversity we have to content ourselves with 'working definitions' of forest types. The term *tropical dry forest*, as defined by Holdridge, (1967) was used for the forests in which the assessments took place, but should I have used the competing terminology *selva baja caducifolia* defined by Rzedowski (1981) that is more generally preferred in Mexico? Whilst I am in these forests, I have to recognise that it is continuum of subtle change, not a sharp distinction, which separates tropical dry forest from evergreen riverine forest. Rarely, however, is such ontological uncertainty acknowledged explicitly in the immutable mobiles with which biodiversity scientists seek to influence the world.

The direct assessment of tree diversity in a Mexican forest is very much a field science, it is performed in an environment very different to that of a laboratory. Much of the critique of sciences that has emerged from the social sciences over the last decade has concentrated on the practices of laboratory based sciences (e.g.

Stengers 2000). Knorr Cetina (1999 p. 27) claims that laboratories give scientists an edge in that

> [N]ot having to confront objects within their natural orders is epistemically advantageous for the pursuit of science.

The precisely controlled environment of the laboratory allows repetition. This is important because one of the rhetorical strengths of the inscriptions that emerge from the laboratory is that each can be put on trial by precise replication in a similar laboratory. However, in the ecological sciences conditions are only very superficially controlled and repetition only imprecisely possible. With Pedro and Jorge we attempted to order our environment by laying out temporary plots to control area and number of trees sampled, and with reasonable accuracy our Geographic Positioning System told us the position of these plots. However, control over our environment could never match that of the laboratory-based scientist. It is improbable that, should we wish to repeat our work, we could relocate those plots with absolute accuracy. Inaccuracy would result in some trees falling into our plots that had previously been outside of them, whilst others would be omitted. Even if we were able to relocate the plots with absolute precision, we would find that in the interval between our first and second measurement a few trees would have died and others would have reached the minimum trunk diameter required for inclusion. They would not be the same plots. Human error enters our work, towards the end of each day we are tired and thirsty, fed up with being bitten by insects and ready to leave, undoubtedly trees are occasionally misidentified under these conditions and some plots will not be measured out perfectly. Our identifications are depended on current taxonomic knowledge and opinion which are both prone to change, species are regularly re-delimited and hence different Linnaean binomials become applicable- *Amphipterygium adstringens* has also been known as *Juliana adstringens*. What was once one species will, under revision, become two, and *vice versa*, whilst whole knew species emerge from the forest that were previously 'unknown to science'; species uncertainty thus presents a serious challenge to the biodiversity assessor (Isaac *et al.* 2004). We can only sample forests- complete inventories are prohibitively costly so our estimated results are framed by the statistical probability that they are reasonably close to the true figure that we will never know precisely. The same protocol will give a different result when reapplied to the same forest. In short, our work is unrepeatable, despite our best efforts to lay out plots, enlist taxonomic specialists in our work and our determination to minimise statistical error.

We cannot boast the epistemic advantage of the laboratory scientist and our version of reality has to be seen is one that is temporally and spatially fixed. Given all these problems, as well as the remoteness of our field site, the validation of our results by peer repetition is a practical impossibility. Even more so than for laboratory based sciences, validation is determined socially rather than by correspondence to an independent reality (Demeritt 1998). Peer review, trust (as much in scientists as in their institutions) and respect for the norms and traditions associated with the analytical techniques we chose are what lend authority to our representations.

Yet once our representations of these forests become cemented into publications such epistemic concerns are hidden or rapidly forgotten and unlikely to be challenged. We become 'experts in our field' and assume the rhetorical advantage that goes being a biodiversity scientist.

4.6 From the Circulation of Names to the Circulation of Publications

Whilst scientific names and representations keep the residents of Santa María Huatulco locked outside of the networks through which the politics of biodiversity conservation work, different circumstances apply to the NGO *Naturaleza*. All the senior members of staff of *Naturaleza* are university educated, and like most other Oaxacan NGOs involved in conservation, count amongst that staff a significant number of biologists. It might therefore be assumed that they would be much better connected to the networks within which biodiversity scientists operate. This appears not to be the case as the literature on conservation biology is largely inaccessible to Oaxacan based organisations. There are no library resources in Oaxaca comparable to even a modest university library in Europe or North American, never mind that the relevant literature is primarily in English and that hard pressed NGO staff would be unlikely to be able to find sufficient time to keep abreast of it were it available[7].

The case of the work of Irma Trejo and Rudolfo Dirzo, two scientists working at the Universidad Nacional Autónoma de México, serves to illustrate the breach between

[7] CONABIO, a semiautonomous federal institute based in Mexico City has increased the quantity of information available on Mexican biodiversity through its web site. The Jardín Etnobotánico de Oaxaca has begun to assemble a modest biological library.

biodiversity scientists and locally based conservation practitioners such as *Naturaleza*. In 2002 they published an important paper entitled *Floristic diversity of Mexican seasonally dry tropical forests* in the respected conservation journal *Biodiversity and Conservation* (Trejo & Dirzo 2002). The paper presents and discusses an assessment of twenty dry forests from across Mexico, and prioritises each forest according to its conservation value. They enacted a performance of biodiversity whose culmination was a scientific paper, an immutable mobile that attempted to represent all the tropical dry forests of Mexico and hence influence resource allocation decisions pertaining to those forests. In this way they resemble Stenger's (2000 p. 128) 'mobilized scientists' who see themselves as 'legitimate representatives of a problem'.

One of the areas that they sampled, 'Copalita', is from the municipality adjacent to Santa María Huatulco, and is therefore from essentially the same forests as those in which *Naturaleza* work. The results of Trejo and Dirzo's paper would be expected to be of some interest to *Naturaleza*. Yet, until I, a UK based researcher, passed them a copy they were totally unaware of the existence of the work. Why should this be?

It is worth reconstructing the pathways along which various representations of these forests must have circulated before this paper came into being. At some point prior to 2002, at least one of the authors must have visited the coast of Oaxaca to carry out the sampling and collecting needed to inform the analysis. In the forests along the Copalita River they would have presumably used some local help for setting out plots and to assist in identifying trees, at least by their local name. Thus the representation of this forest would have begun much as described above. Data would have first been noted in field books before transfer to computer, whilst botanical collections would have been pressed before transfer to MEXU where they would have had their scientific names determined prior to labelling and mounting. Together these collections form dispersed representations of the twenty forests, hidden amongst the other million specimens in MEXU. However, none of the community of Oaxacan conservation NGOs were directly involved in the execution of this work.

At some point, when sufficient specimens had been identified, a second representation of the Copalita River forest would have begun to emerge from the computer analysis of the data. It is this representation that would ultimately circulate in the publication, via a European publishing house, through the biological and environmental departments of the world's better funded universities, finally coming to my attention through my privileged access to the information technology service of a high income country university. It did not however reach the people most likely to act upon its findings. When I spoke to the staff of *Naturaleza* they had no knowledge of the paper despite knowing of the authors. Paradoxically, the advances in communications that cause a kind of space-time compression for those in a position to make use of them tend to isolate those who are not (Massey 1991). Subscriptions to scientific journals are expensive, and scientists often consider their work done once their representations are circulating in academic networks. Such work exists deliberately separate from the social context in which it was produced, separate from other representations of the forests and trees and thus decontextualised from many local concerns. Indeed the biodiversity scientist with remote access to her northern research institute's information technology services can access more information about Oaxacan biodiversity from an internet café in Oaxaca than can the staff of *Naturaleza*. Hence Oguibe's (2002 p. 177) concern that the internet:

> [P]rovides a new corridor of infringement and trespass which the infringed may not always be privileged to broach.

Yet so often it appears that this is exactly the scenario that biodiversity science aspires to, nature as a single representation, circulating in highly exclusive networks. The opposite of the 'connected and responsible' science Latour (1999 p. 97) hopes to achieve.

4.7 Discussion

This chapter has attempted to show that the language and inscriptions used in the assessment of biodiversity, 'the living nature of the contemporary western biologist' (Soulé 1995), are culturally situated and not inclusive to many of the actors involved in Oaxacan biodiversity conservation.

Biology and politics have long been intimately associated; the questions biologists ask being shaped by both nature and politics (Haraway 1998). In turn, the immutable mobiles of scientific representation have power, but that power is not derived from their mobility *per se*, as it has been shown other types of representation are at least as mobile, nor is it justified by a superior claim to objectivity. Instead their power comes from their privileged access to the networks of the epistemic community.

The standardised nomenclature, and the accepted procedures and analysis of biodiversity assessment, all have their particular positionalities as have the local representations of Oaxacan dry forest trees with which they compete. Scientific representations of nature are still far from being a universal *lingua franca* for nature and as Escobar (1999 p. 5) contends:

> It is well accepted already that nature is differently experienced according to one's social position and that it is differently produced by different groups or in different historical periods.

Whilst there is, therefore, a constructivist interpretation of representations of biodiversity, it is important to clarify that this is not an attempt to show that biodiversity is simply willed into existence in the minds of scientists. Biodiversity is both a product of the biodiversity scientists' positionality and an interpretation of a more widely shared reality. In this context there is much to recommend the *artefactual constructivism* of Demeritt (1998) that is ontologically realist but epistemologically anti-realist. What both *cuachalalate* and *Amphipterygium adstringens* represent is an entity in a forest recognised consistently by both me and Pedro, even if the ways in which we account for it are different. Or as Latour (2000 p. 119) argues, 'things', in which I will include biodiversity, are:

> [M]uch too real to be representations, and much too disputed, uncertain, collective, variegated, divisive to play the role of a stable, obdurate, boring primary qualities, furnishing the universe once and for all.

Here it has been shown that biodiversity assessments are achieved by the articulation of a set of social, historical and biological relations that continue to perform as they travel. The problem appears to be that Oaxaca, where locally situated representations continue to dominate, is left behind.

To suggest that the divergence between these local and scientific accounts of nature amounts to a clear dichotomy is overly simplistic. As will be shown in chapter 8, local NGOs occupy an ambiguous role in the shift between the two (Leach &

Fairhead 2000). However, given this imperfect translation, it is hardly surprising that biodiversity assessment is not rigorously applied in Oaxacan conservation.

Here the idea of 'hybidity' (Escobar 1999; Whatmore 2002) is useful in that it allows us to work between the polarized extremes of the unhelpful realist versus relativist debate on nature and move on to more immediate concerns. The fragment of Mexican nature variously known as *cuachalalate* and *Amphipterygium adstringens* shows many of the properties of hybridity described by Whatmore (1999). Even the forest from which the tree once came cannot be said to be purely natural, these forests, perhaps like all others (Denevan 1992; Gómez-Pompa & Kaus 1992), are most certainly not untouched by human hands. Since prehispanic times they have been cleared, farmed and allowed to regrow. Cattle and goats now graze them, and precious timbers have been removed. The nature which we attempted to assess for its biodiversity is thus:

> [A] relational achievement spun between people and animals, plants and soils, documents and devices in heterogeneous social networks which are performed in and through multiple places and fluid ecologies. (Whatmore 2002 p. 37).

The appeal to biologists of biodiversity is that it is a nature that is tractable, countable and amenable to quantitative analysis, it is nature excised from the culture(s) in which it has to be conserved. It is necessarily a simplistic reading of nature that cannot reach all of Whatmore's (ibid) 'multiple places'.

Why then is this so rarely acknowledged by biodiversity scientists? Possibly it is because the exclusive networks through which their representations travel do not question the fundamental premises of their work. Golinski (1998 p. 29) argues that:

> Most scientists, most of the time, live their lives within a supporting matrix of trust. It is only when that trust is broken down that the social mechanism is exposed to view.

The point is not to question the legitimacy of biodiversity and its representations, but to question how they have come to have hegemonic status in the political agenda that promotes nature conservation and further, to investigate how that agenda is played out in Oaxaca. In other words, realism is not the issue; articulation is (Latour 1999). Described in the starkest terms, the context of biodiversity research in

Oaxaca is one in which 'outside' scientists, operating in privileged networks appropriate information about local biological resources. As such biodiversity assessment can be seen as a manifestation of the postcolonial condition. Bhaba (1994 p. 171) states that the postcolonial critique:

> Bears witness to the uneven forces of cultural representation involved in the contest for political and social authority within the modern world order.

Biodiversity assessment, perhaps unwittingly, contributes to this unequal representation because the discourse of biodiversity conservation is so exclusive, ensuring that decisions about the periphery (e.g. Santa Maria Huatulco) are taken at the centre (e.g. Mexico City and beyond). It is precisely this that makes the ways in which nature is represented such a concern.

Young's (2003) argument that much of the postcolonial struggle has been, and is about, the control and distribution of land is pertinent here. The inevitable conclusion that is to be drawn from much research on biodiversity is that certain areas should be given protection and others not. This can entail dramatic changes in local land use politics, including increased regulatory powers being enforced from outside. This concern is exacerbated by the accident of biogeographical history that has resulted in many of the places biodiversity scientists deem most in need of protection being found at the 'periphery', in the rural areas of lower and mid-income countries. It is not surprising that conservationists, amongst whom we can include biodiversity scientists, often find themselves cast as reactionaries in a post colonial world (e.g. Guha 2000; Vidal 2001). As a result it is likely that biodiversity scientists will increasingly be forced out of the 'supporting matrices' that Golinski described and have to confront open debate about the politics of their representations.

The reclamation of what Gandhi (1998) calls non-European knowledges, may go some way to addressing this imbalance between centre and periphery. However, given the inherent complexity of scientific nomenclature and inscriptions of biodiversity, it is perhaps overly optimistic to expect the residents of Santa María Huatulco to engage scientists on their own terms anytime soon. Levels of education and availability of information currently make this unlikely. Whilst assessment protocols that pay greater attention to local knowledges are now at least being discussed, (e.g. Vermeulen & Koziell 2002; Wong *et al.* 2002) it seems likely that biodiversity scientists will continue to require assessments based on 'their terms'.

We might attribute this to their desire to maintain their position of dominance in conservation politics or to the practical need for a universal denominator (i.e. biodiversity) in the project of conserving global nature, if indeed these two reasons are not different sides of the same coin. Thus the 'imposed hierarchy', which Bhaba (1994 p. 4) hopes might be overcome by a more hybrid approach to culture and nature, is likely to prove resilient for some time yet.

Having therefore established that biodiversity assessment does not attempt to capture the diversity of cultural relations that nature and humans, it is both necessary and justifiable to consider it on its 'scientific merits'. The following two chapters therefore ask how well biodiversity assessment, as currently practiced, meets it objectives for describing dry forest tree diversity, and how it might be improved.

Chapter 5: Assessing woody diversity in a tropical forest-Local variation and its effect on regional scale assessments

5.1 Introduction

In the initial stages of the planning and implementation of a biodiversity conservation initiative, priority setting is essential if the maximum return is to be gained from the investment of the limited resources available (Pressey *et al.* 1993). Biodiversity assessment [which here is taken to be synonymous with biodiversity inventory (Stork & Samways 1995)] methods may be used to locate and identify important biodiversity such as the locations of threatened or endemic species and threatened or speciose habitats. At global and regional scales, international conservation organisations have carried out a variety of such assessments, including Conservation International's Global Hotspots (Myers *et al.* 2000), Birdlife International's Endemic Bird Areas (ICBP 1992), the WWF/IUCN's Centres of Plant Diversity (Davis *et al.* 1997) and WWF's Global 200 Ecoregion Assessments (Dinerstein *et al.* 1995).

By necessity, global scale and regional scale biodiversity assessments have to rely on a variety of primary and secondary information sources that vary in quality and quantity depending on the areas assessed. Expert analysis is therefore a crucial part of such prioritisation exercises, with location-specific and/or taxon-specific expertise being called on to collate, weigh and adjudicate available information. The results of such assessments represent the best understanding of biodiversity priorities at any given time, and despite the shortcomings in the assessments undertaken to date, they are being used as a basis for targeting conservation resources (Redford *et al.* 2003).

The fact that these assessments depend heavily on expert analysis has led to increasing focus on quantitative methodologies for the direct measurement of biodiversity. However, because of variation in the amount to which different localities have attracted the interest of biodiversity scientists and the degree to which expert opinion is required to facilitate interpretation and comparison of data, strict comparison between locations is not easy. It is left to the experts to decide to what extent differences between the areas assessed result from real differences or are artefacts of differing target biota and sampling methodologies. Perhaps the greatest concern is that under-sampled areas are likely to be over-looked completely.

At a local scale, plant biodiversity inventories, particularly at the species level, have generally relied on standardised field techniques to facilitate comparison (e.g. Hall 1991; Phillips *et al.* 2003; Stern 1998; Vanclay 1998). The speed and efficiency of these techniques has led them to become known generically as rapid biodiversity assessment (RBA) techniques and has allowed them to be applied not only at local scales, but to be repeated across regions and thus be used to investigate biogeographical patterns (e.g. Gentry 1995). However, for valid comparisons across regions to be made, samples of local vegetation must adequately represent local diversity. Therefore, in the execution of regional scale assessments, consideration must be given to how local scale sampling is carried out.

This issue is highlighted by two recent studies from Mesoamerica. Trejo and Dirzo (2002) compared tropical dry forests across Mexico from the north west of the country to the south and south east, whilst Gillespie *et al.* (2000) compared dry forests from western Costa Rica and Nicaragua. In both of these cases the various localities compared were represented by single sites. This appears also to be the case for assessments of many forest types across the globe; Phillips *et al.* (2002) compiled surveys based on one or a few sites per locality from a variety of tropical forest types and regions worldwide. What is not clear is whether these single or few sites adequately represent each locality from which they come or whether neighbouring forests within the locality are significantly different from those sampled. There is therefore a need to consider whether single small samples from geographically distant locations can be meaningfully compared at regional scales, and hence whether the conservation prioritisations based on such sampling approaches are valid. Comparison of biodiversity patterns at different scales is further hindered by terminological confusion. In the following, a *locality* refers to an area of a size suitable for consideration as a single conservation management unit. Regional scale assessments are therefore comparisons between several such localities. A *site* is a relatively homogenous patch of forest from which a single *sample* might be taken. A locality is therefore composed of many sites and the assessment of a locality is a local scale assessment. Given the sampling protocol used here, each sample is composed of several *plots*.

The aim of this chapter is to test whether sampling a single site from a locality, as typically carried out for regional rapid biodiversity assessments, results in adequate representation of the diversity of that locality for the purposes of comparison with other localities elsewhere in the region or world. This question is addressed here with reference to patterns of local and regional diversity in tree species of Mesoamerican tropical dry forest of the lowlands of the Pacific watershed of Mexico and Central America. Consideration is given to overall diversity as well as diversity of restricted range species.

The Mesoamerican dry forest is a global conservation priority (Lerdau *et al.* 1991; Mooney *et al.* 1995). Although it is likely that sampling issues dealt with here will be pertinent to any biome where significant edaphic, topographic and climatic variation is found, the need for refocusing on the local scale in the Mesoamerican dry forest biome is particularly important because of its highly fragmented nature and the diversity of landscapes that exist within it. As a result of this natural and anthropogenic variation, considerable variation may occur in vegetation across scales measured in hundreds or thousands of metres, whilst regional analyses more typically seek to differentiate patterns across scales measured in hundreds or even thousands of kilometres.

5.2 Methods

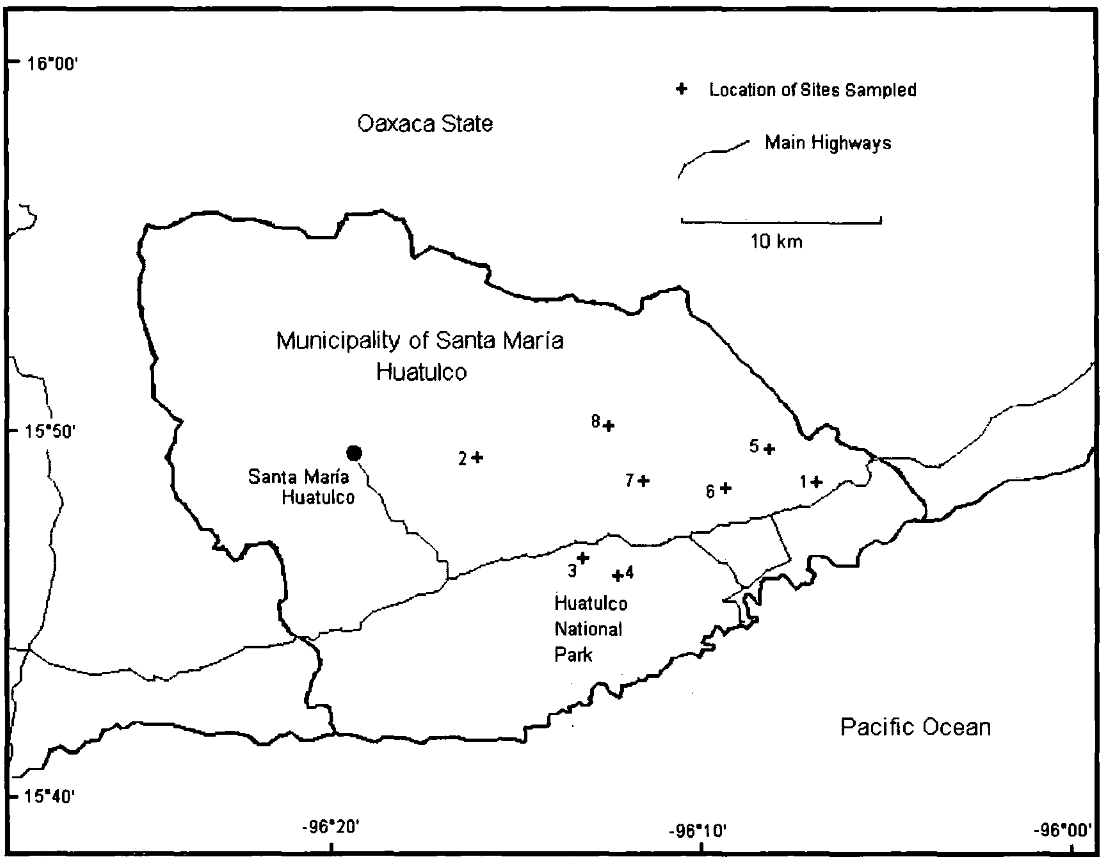
5.2.1 Introduction

Spatial variation in patterns of tree species diversity in tropical dry forests at one locality in southern Mexico were assessed and described by taking samples from eight sites. The results of this intensive local assessment were then compared to patterns described by two previously published regional scale assessments across Mexico (Trejo & Dirzo 2002) and Central America (Gillespie *et al.* 2000). The field protocol chosen for the local assessment is a variation of the 0.1 ha rapid assessment technique popularised by Gentry (1982) and used in these two regional assessments.

5.2.2 Study Area

All of the forests sites sampled fall within a single locality of approximately 200 km^2 in the Municipality of Santa María Huatulco (henceforth referred to as Huatulco) on the coastal lowlands of Oaxaca. This municipality contains 6000 ha of federally protected forest in the Huatulco National Park and smaller communally protected areas managed by the municipal and communal authorities in collaboration with a local non-governmental organisation. Besides tropical dry forest, the municipality also contains extensive areas of semideciduous forest as well as some areas of mangrove and savannah, but dry forest is the exclusive concern here. The forests sampled in this area (figure 5.1) all conform to the definition of dry deciduous woodland (*bosque seco cauducifolia*) of Rzedowski (1981), that is, closed canopy broadleaved forest, with a canopy that is rarely higher than 10 m and that is strongly drought deciduous.

Figure 5.1 Location of Eight Dry Forest Sites Sampled in Santa María Huatulco, Oaxaca, Mexico.



The vegetation of the locality falls within the 'Balsas dry forest' ecoregion of Dinerstein *et al.* (1995) and is similar to the 'Copalita' sample of Trejo and Dirzo (2002) which is estimated to be about 10 km to the east of Huatulco. Annual rainfall is 1127 mm, of which 95% falls between May and October (INEGI 2001). Within the locality, tropical dry forest is distributed as fragments ranging from 50 ha to 1000 ha in size, concentrated in the south and east of the municipality at altitudes of 10 to 150 m.a.s.l. Soils throughout the locality are acidic regosols (Álvarez 1998) and the topography is undulating with occasional rocky outcrops. The plant diversity of the dry forests of this region has recently been extensively check-listed by Salas *et al.* (2003).

The criteria used for selection of a site were that it should be closed canopy dry forest within the municipality and that access could be gained to it from the relevant authority or owner. Prior to commencing fieldwork a list of 15 potential sites was drawn up using information from local informants. These 15 sites displayed a range of successional states and degrees of anthropogenic disturbance. From these sites eight were selected randomly for the assessment described here.

5.2.3 Survey Methodology

The methodology used sampled each site with 10 plots of 2 x 50 m. The principal aim of this research was to test whether rapid sampling at a single site can represent diversity in the wider locality, not to provide a critique of this particular protocol. The method was chosen not because it is necessarily considered superior to other sampling protocols, but because it has become one of the most common methodologies used by plant diversity assessors working in forested landscapes in tropical regions (Phillips *et al.* 2003). Unlike Gentry (1982), lianas were excluded from the assessment and the lower diameter limit for inclusion of trees and shrubs was set at 5 cm. For comparative purposes an additional 5 plots were surveyed at each of the eight sites selected in the locality. Each of the 2 m wide plots was then extended to 6 m width and re-assessed. This enabled the results from samples of ten 2 x 50 m plots (0.1 ha total area), as typically used in Gentry's methodology, to be compared with the results of samples of fifteen plots of 6 x 50 m (0.45 ha total area).

Plots within each site were located randomly, with respect to a 'base line' formed by the trail followed in order to enter the forest. The majority of trees encountered were identified in the forest, but for those which were not identifiable, or whose identity was considered in anyway doubtful, voucher specimens were taken and deposited in the QUIE community herbarium in Oaxaca for later identification with the aid of expertise from the National Herbarium (MEXU) in Mexico City. Vouchers that could not be confidently identified to species were treated as morphospecies in the analyses below.

5.2.4 Species Weighting

The samples were initially compared without weighting of their component species, thus the analysis reflects comparative species richness. Then species were weighted according to risk of extinction, approximated in inverse proportion to range size, thus facilitating an analysis that reflects differences in between-site bioquality *sensu* Hawthorne (1996). Such weighting depends on identification of the species and morphospecies were excluded from this part of the analysis. Restricted range is used here as a surrogate of extinction risk, that is, the assumption is made that species occupying less of the earth's surface are more likely to go extinct than those that occupy a greater area. As Gaston (2003 p. 91) points out, this relationship is probably real but little tested.

The weighting used estimated the range size of each species based on herbarium specimen information held at MEXU and in the *Tropicos* database of the Missouri Botanic Gardens (w^3Tropicos undated). Given the variable precision with which herbarium specimen localities are recorded, the number of Mexican states/Central American countries in which a species had been collected was used to estimate distributions. The first step in creating a weighting was to calculate a relative area for each state and country by dividing the land area of each of those political entities by the land area of Oaxaca (thus giving Oaxaca a relative area of 1). To create a score for each species, the total relative land area of the states/countries in which each species was found was calculated and inverted. Thus the highest score obtained for any species was 1 (for Oaxacan endemics) with all other species, which must be known from Oaxaca and at least one other state/country, scoring greater than 0 and less than 1. In this way the species of most restricted range were given the highest weighting. Gordon *et al.* (2004) classified approximately 20% of the Mesoamerican dry forest species they identified in this region as being of conservation concern and here the most restricted 20% of species were selected for analysis of bioquality.

5.2.5 Analysis

A single species accumulation curve was constructed for the eight sites combined and for both sampling protocols (0.1 ha and 4.5 ha). Estimates of total species in this landscape were calculated using a range of non-parametric indicators (ACE, ICE, Chao 1, Chao 2, Jackknife 1, Jackknife 2 and Bootstrap: Colwell 2004b), to indicate the percentage of species on which the assessment of this locality is based. This was repeated for the restricted range species.

Of particular relevance here is the comparative variability in species diversity between the Huatulco assessment and the regional assessments of Gillespie (2000) and Trejo and Dirzo (2002). In these regional assessments, the lower diameter limit for inclusion of trees was 2.5 cm, whilst here 5 cm was used. In order to compare the range of estimated total species from each of these studies the total number of species at each site was expressed as a percentage of the species number of the most speciose site in each study. The resulting variable is denoted $S_{(\% \text{ of max})}$.

The observed number of species for each site/locality, $S_{(obs)}$, results from a sample and is therefore a point estimate of the true number of species at the site/locality. If these point estimates have large sampling errors, even if they go unreported, then the ranking of localities based on $S_{(obs)}$ would be dubious. The mean and confidence interval of $S_{(obs)}$ in the Huatulco assessment were calculated. The same statistic cannot be calculated for each locality in the two regional assessments of Gillespie *et al.* (2000) and Trejo and Dirzo (2002) because only one site was surveyed at each locality. As an alternate method of accounting for variability within these assessments, the confidence interval calculated for the Huatulco was fitted around the point estimates of each of the localities in the other assessments. Thus it was assumed that had several sites at each of those localities been sampled a similar between site variability might have resulted.

There is now a growing consensus that site selection in conservation should be driven by consideration of the composition of species at a site, not just the comparisons of species richness (e.g. Önal 2003; Rodrigues & Gaston 2002). Therefore the ability of assessment exercises to distinguish between sites and localities by species composition needs to be considered. Species composition between sites in Huatulco was analysed using two complementary analytical techniques. The first was detrended correspondence analysis (DECORANA), a technique based on reciprocal averaging which represents visually and in two dimensions the multi-dimensional relationships between samples made up of similar, but not identical sets of entities (the entities in this case being tree species). Here DECORANA was used for presence/absence comparison of the species composition of each plot at each site, rather than weighting species according to their relative abundance. Presence/absence analysis was preferred as the aim was biodiversity assessment rather than an ecological characterisation of each area sampled. Under these circumstances the presence of a single individual of a species may be of equal significance to the presence of the most abundant species.

Reciprocal averaging techniques such as DECORANA tend to emphasize between-sample differences, regardless of the significance of those differences. Several similarity indices are available to test between-site differences in species composition. Here the Sørenson index was chosen because of its logical simplicity and its use by Gillespie *et al.* (2000) and Trejo and Dirzo (2002).

The Sørenson index is given by:

$$C_s = 2j/(a + b)$$

Where $0 \leq C_s \geq 1$; j = the number of species found in both sites and a and b = the number of species found in sites A and B respectively, (Magurran 1988). A score of 0 implies total dissimilarity, 1 implies complete similarity.

In the following analysis the sample from each site was broken down into its constituent plots with within-site variation being measured by comparison of pairs of plots from within a site, and between-site variation being measured by comparison of pairs of plots from different sites. In this way the independence of within site and between site variation was tested. The distribution of the index thus derived is not normally distributed, but right-skewed due to the excessive number of zero scores obtained. Thus the Mann-Whitney U test was used to test the difference between medians. Note that given the large number of between-plot comparisons, the sampling distribution of U approaches that of the normal distribution, Z, hence the latter is used as the test statistic in tables 1 and 2 (Siegel & Castellan 1988).

DECORANA was carried out using PCordwin™ (McCune & Mefford 1997), the similarity indices were calculated using EstimateS (Colwell 2004b) and the Mann-Whitney U test was performed using SPSS (SPSS Inc. 2001).

Comparisons with other studies were made difficult by the lack of published information on species composition. However, comparisons with the surveys of Gillespie *et al.* (2000) and Trejo and Dirzo (2002) based on species richness, rather than composition, were possible. In this way, variation in species number per locality across Mesoamerica could be compared with variation per site within a single locality in Huatulco, Mexico. The variation in these previously published indices was compared with those of the eight sites sampled in Huatulco using the F-test. It was then assumed that the same variance occurs between sites within the locality of Huatulco as between sites within the localities included in the regional surveys published previously to test the effect of local variation on the those surveys.

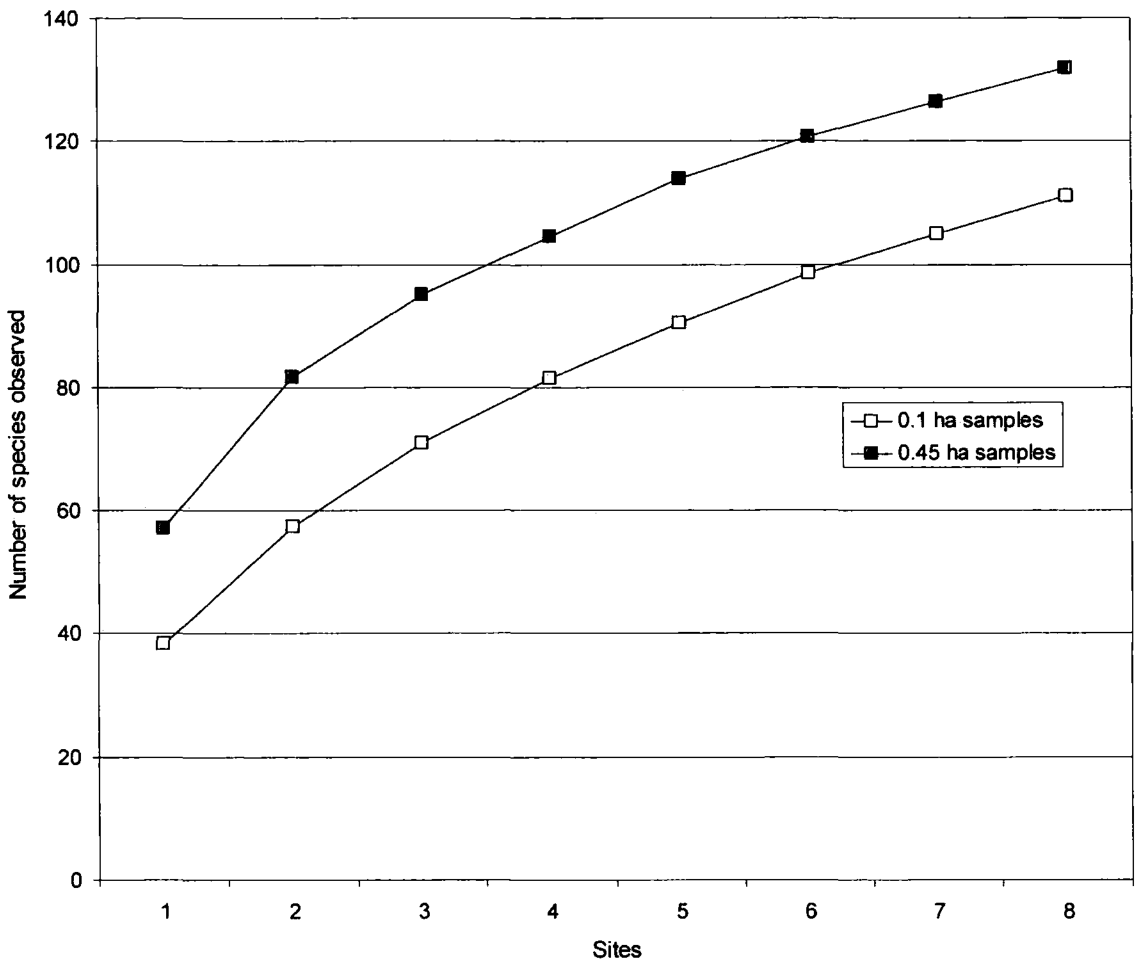
5.3 Results

5.3.1 Species Richness

Estimates of total species

The eight 0.1 ha sampling revealed a total of 111 species of which 10 were morphospecies. The 0.45 ha sampling revealed 132 species of which 12 were morphospecies. Figure 5.2 presents the untransformed species accumulation curves derived from both sampling protocols. It shows that for both protocols, samples from single sites revealed less than half of the tree species (> 5 cm dbh) observed across the locality. For the 0.1 ha locality, prioritisation based on a single site-sampling is based on just 34% of known species, whilst for the 0.45 ha protocol it would be based on 43% of known species. However, even after sampling eight sites, neither curve appears close to reaching an asymptote, suggesting that there are more species to be found at this locality. From the range of non-parametric estimators of species richness used, the highest estimate of species richness in this locality derived from the 0.1 ha protocol was 199 species (Chao 1 estimator) and the lowest was 130 (Bootstrap estimator). These estimates suggest that prioritisation based on a single site sample would be based on between 19% and 29% of total species from the locality. For the 0.45 ha protocol the highest estimate was 188 species (Jackknife 2 estimator) and the lowest was 148 (Bootstrap estimator) thus prioritisation based on a single site sample would be based on between 30% and 39% of total species from the locality.

Figure 5.2 Comparison of species accumulation curves (50 repetitions) for trees in eight tropical dry forest sites in Oaxaca, Mexico using two plot-based sampling protocols.



Estimates of variability in species observed

Table 5.1 Summarized species diversity statistics for three surveys of Mesoamerican dry forests using 0.1 ha sampling protocol. Lower diameter limit for Huatulco samples is 5 cm, for others 2.5 cm. S(obs) = number of species observed at each locality/site; S(% of max) = S(obs) expressed as a percentage of the most speciose locality/site. CI(est) is the estimated 95% confidence interval of S(obs) for each locality derived from between-site variation in the Huatulco site survey. Data are presented in rank order, with the most speciose locality/site given first.

	Mexican localities (n = 20)*			Central American localities (n = 7)**			Huatulco sites (n = 8)	
	S(obs)	S(% of max)	CI(est)	S(obs)	S(% of max)	CI(est)	S(obs)	S(% of max)
	90	100.0	12.1	75	100.0	12.1	42	100.0
	76	84.4	10.2	65	86.7	10.5	41	97.6
	75	83.3	10.1	59	78.7	9.5	40	95.2
	73	81.1	9.8	54	72.0	8.7	40	95.2
	63	70.0	8.5	48	64.0	7.7	39	92.9
	63	70.0	8.5	45	60.0	7.3	36	85.7
	58	64.4	7.8	44	58.7	7.1	33	78.6
	58	64.4	7.8				26	61.9
	53	58.9	7.1					
	52	57.8	7.0					
	52	57.8	7.0					
	50	55.6	6.7					
	44	48.9	5.9					
	42	46.7	5.7					
	41	45.6	5.5					
	41	45.6	5.5					
	35	38.9	4.7					
	34	37.8	4.6					
	33	36.7	4.4					
	22	24.4	3.0					
mean	52.8	56.4		55.7	70.0		37.1	86.7
SD	17.1	16.8		11.4	11.7		5.4	12.8

Table 5.1 summarises diversity statistics for two regional assessments (Mexico and Central America) and one local assessments (Huatulco) of Mesoamerican dry forests. Huatulco was the least diverse as measured by mean $S_{(obs)}$, which may at least in part be attributed to the fact that this survey employed a higher minimum diameter than was used in the other investigations. The range of $S_{(\% \text{ of max})}$ is greatest for the assessment of Mexican localities, and least for the Huatulco sites. Whilst the range of $S_{(\% \text{ of max})}$ for Huatulco is approximately half that for the Mexican assessment, the eight most speciose sites from the latter, between-locality assessment fall within the range of 100 to 61.9% calculated for the between-site assessment of Huatulco. The range of $S_{(\% \text{ of max})}$ for all the Central American localities is only slightly greater than that of the Huatulco sites.

For the Huatulco sites the 95% confidence limit of the eight estimates of $S_{(obs)}$ was ±4.48. Expressed as a percentage of the mean value (37.1) this is 12.1%. When this interval, $CI_{(est)}$, is fitted to each of the localities from the regional assessments, the result is considerable overlap in 'estimates' of $S_{(obs)}$ that would result in alteration of between-location priorities (table 5.3). For the Mexican assessment, only the most and least speciose localities have a $CI_{(est)}$ that does not overlap the total species count of at least one other locality, and the lower bound of the $CI_{(est)}$ of the most speciose locality overlaps the upper bound of second most speciose locality. For the Central American localities, there is no $S_{(obs)}$ that does not fall within a $CI_{(est)}$ of at least one other locality. An extreme example further illustrates this point: the ninth most speciose locality in the Mexican assessment ($S_{(obs)}$ = 53) has a $CI_{(est)}$ large enough to place this site above the seventh most speciose locality or below the twelfth most speciose locality.

5.3.2 Species Composition

Table 5.2. Sørenson's index for eight dry forest sites assessed using two 0.1 ha and 0.45 ha sampling.

Site	0.1 ha samples							0.45 ha samples						
	2	3	4	5	6	7	8	2	3	4	5	6	7	8
1	0.52	0.59	0.49	0.55	0.55	0.27	0.44	0.63	0.64	0.60	0.72	0.64	0.42	0.61
2		0.55	0.49	0.45	0.54	0.42	0.53		0.69	0.61	0.60	0.66	0.56	0.71
3			0.58	0.48	0.44	0.36	0.48			0.66	0.63	0.62	0.57	0.72
4				0.42	0.40	0.41	0.36				0.56	0.54	0.47	0.70
5					0.40	0.28	0.41					0.63	0.43	0.63
6						0.35	0.49						0.47	0.67
7							0.30							0.50
						Mean	0.45						Mean	0.60

The Sørenson's indices (table 5.2) show that all between-site comparisons were lower for the less intensive inventory protocol than the more intensive protocol, with the difference between the two medians being significant (Mann-Whitney U test, $p < 0.01$). This suggests that the degree of similarity between sites sampled by the 0.1 ha methodology is at least in part due to under-sampling of the sites, as with more intensive sampling the between-site similarity increases.

Figure 5.3. Detrended Correspondence Analysis (presence/absence) of eight tropical dry forest sites in Oaxaca, Mexico based on 0.1 ha (ten 2 x 50 m plots) samples.

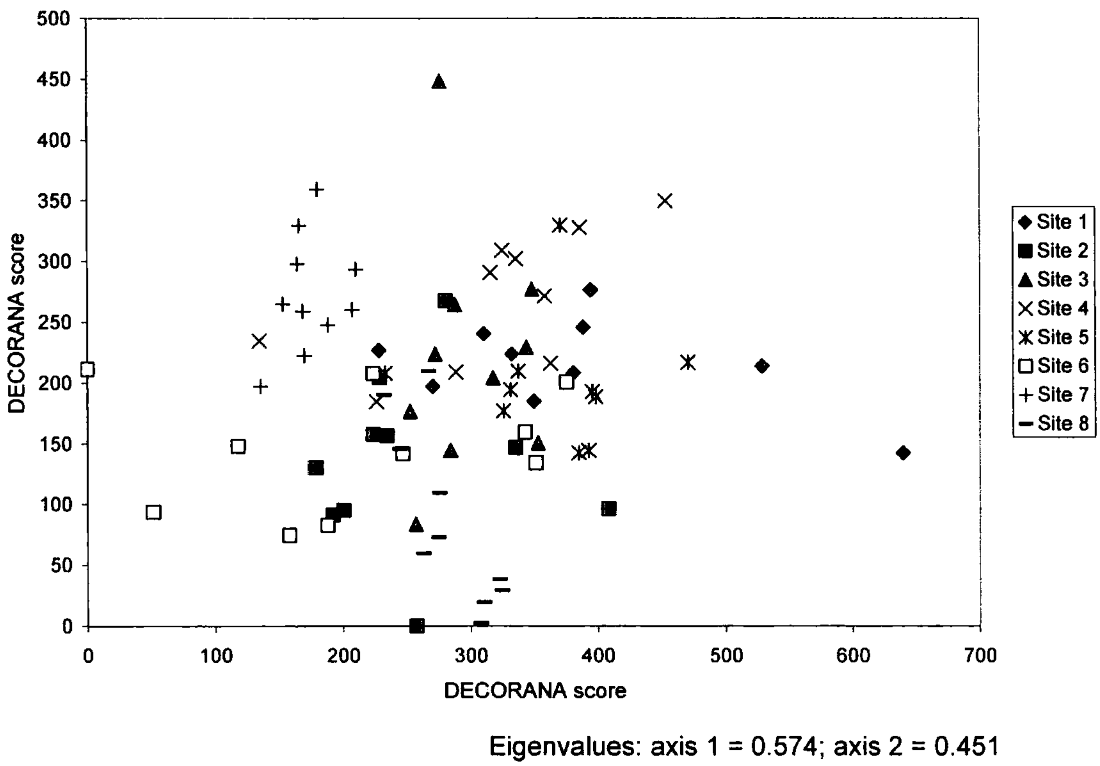


Figure 5.3 shows the relationship between the species composition of each of the ten 2 x 50 m plots across the eight sites surveyed, with plots of more similar species composition being clustered closer together. This clustering was most notable for sites 5, 7 and 8, and suggests that, whilst there is overlap in composition between sites, differences in within-site diversity are less than differences between sites (i.e. α-diversity < β-diversity).

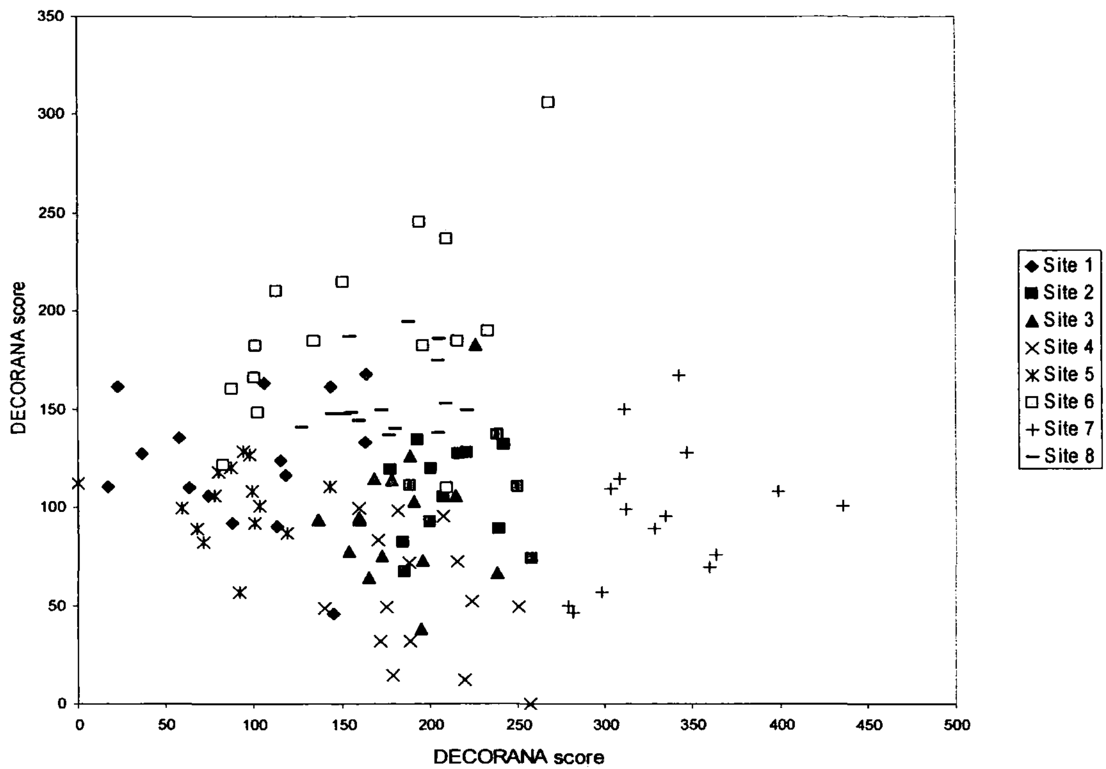
Table 5.3. Mean within-site and between-site indices of similarity for samples of 0.1 ha (ten 2 x 50 m plots) of tropical dry forest in Huatulco Mexico. Within-site similarity measured as the mean of all pairs of plots within a site, between-site similarity measured as the mean of all combinations of pairs of plots from different sites.

	Within site								Between sites
Site	1	2	3	4	5	6	7	8	
Mean species/ plot as % of site	17.9	19.7	20.3	20.3	19.3	18.9	23.3	20.7	-
Sørenson's index	0.11	0.17	0.20	0.18	0.20	0.18	0.30	0.21	0.12

Table 5.3 shows that for all but site 1, within-site similarity is higher than between-site similarity (i.e. α-diversity < β-diversity). This suggests that sampling at any single site is unlikely to capture total variation across the locality. These differences are highly significant (Mann Whitney U test, all within-site scores against all between-site Sørenson's scores, $Z = -10.336$, $p < 0.000$).

This evidence alone is, however, not sufficient to reject the possibility that a sample taken from a single site can adequately capture diversity across a locality. It may be that the sampling protocol employed did not adequately capture diversity at a single site and therefore that more intensive sampling at any given site might result in an estimate of diversity at one site that is not significantly different to that of the locality as a whole. This was tested by using the second sampling protocol describe above in which the area sampled per site was increased to 0.45 ha. The DECORANA results of this sampling are shown in figure 5.4.

Figure 5.4 Detrended Correspondence Analysis (presence/absence) of eight tropical deciduous forest sites in Oaxaca, Mexico based on 0.45 ha samples (fifteen plots of 6 x 50 m).



Eigenvalues: axis 1 = 0.400; axis 2 = 0.263

Visual inspection of figure 5.4 reveals that more intensive sampling has not reduced clustering of plots from single sites, i.e. β-diversity appears still to be high. The clustering of plots taken from each site is in fact more marked than in figure 5.3. It appears that capturing a greater proportion of within site diversity does not necessarily make any one site more representative of the others.

Table 5.4. Mean within-site and between-site indices of similarity for samples of 0.45 ha (fifteen 6 x 50 m plots) of Mexican tropical dry forest. Within-site similarity measured as the mean index score of all pairs of plots within a sample, between-sample similarity measured as mean index score of all combination pairs of plots from different sites.

	Within site								Between sites
Site	1	2	3	4	5	6	7	8	
Mean species/ plot as % of site	23.0	29.9	25.6	27.2	26.9	22.1	24.6	24.8	-
Sørenson's Index	0.33	0.43	0.36	0.36	0.38	0.30	0.43	0.37	0.22

Table 5.4 shows that both within-site similarity and between-site similarity increased with greater sampling intensity; within-site similarity remaining greater than between site similarity. These differences are highly significant (Mann Whitney U test, all within-site scores against all between site scores, $Z = -26.353$ $p < 0.000$). This shows that under more intensive sampling, within-site diversity is not equivalent to between-site diversity. Figure 5.3 illustrated this finding in a different way. Whilst increasing the area sampled by a factor of 4.5 increased the number of species found at a each site this still resulted in less than half the known diversity of the locality being detected.

Species composition

Table 5.5. Summarized Sørenson indices for three surveys of Mesoamerican dry forests using 0.1 ha sampling protocol. Each column summarises the between-site percent similarity for all combinations of two sites in each survey. Lower diameter limit for Huatulco samples is 5 cm, for others 2.5 cm. Max S is the species count of trees from the most speciose site in each the survey, min S is from the least speciose site in each survey.

	Mexican localities (number of combinations = 192)*	Central American localities (number of combinations = 21)**	Huatulco sites (number of combinations = 28)
max	0.46	0.51	0.59
min	0	0.16	0.27
mean	0.09	0.30	0.45
SD	0.08	0.10	0.09

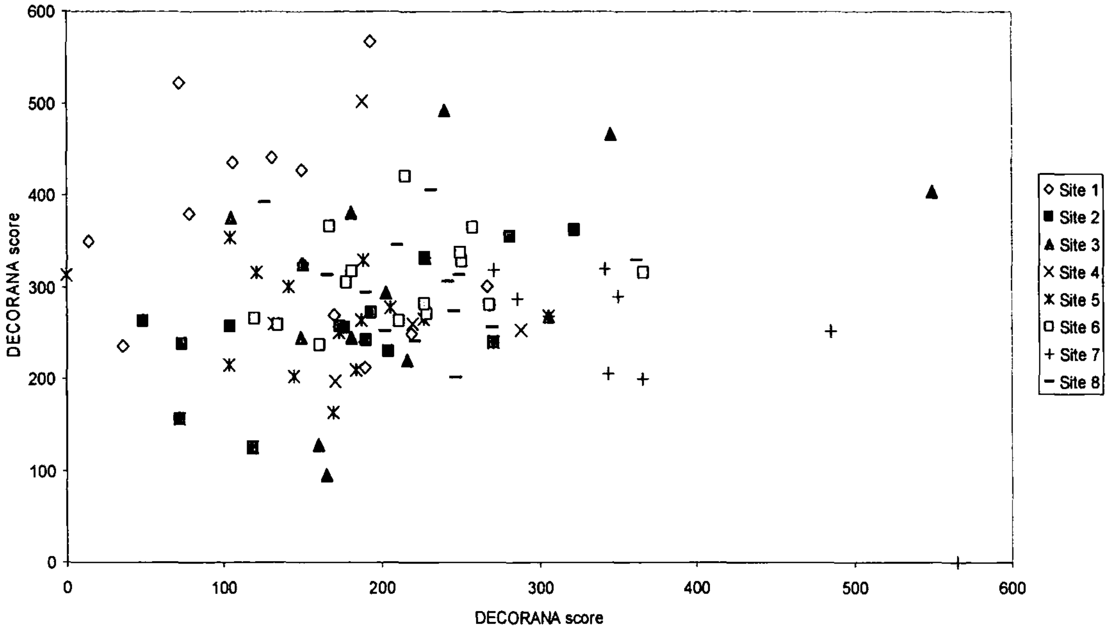
* from Trejo and Dirzo (2002), ** from Gillespie *et al.* (2000)

As would be expected, Table 5.5 shows that there is less similarity in species composition, indicated by lower indices of similarity, between sites taken from various localities ('Mexican' and 'Central American') than there is from sites from within a locality (Huatulco). However, variation within these different surveys (measured by the standard deviation) is similar and not found to be statistically different (F-test of the variance, Huatulco sites v Mexican localities and Huatulco vs Central American localities $p > 0.05$ in both cases). The implication of this, if Huatulco is typical of dry forest localities, is that within-locality variation is likely to be an important contributor to between-locality variation. As a measure of variability of composition, the 95% confidence intervals of the mean Sørenson's index for the Huatulco sites is 0.34, or 7.6% of the mean. This is less than the 12.1% calculated for estimates of species richness.

5.3.3 Restricted Range Species

Here the focus of analysis is the subset of the total species that are considered threatened because of their restricted ranges. In order to maximise representativity, data from the more intensive 0.45 ha protocol is used.

Figure 5.5 Detrended Correspondence Analysis (presence/absence) of eight tropical dry forest sites in Oaxaca, Mexico based on 0.45 ha samples, with only the 20% most restricted range species included.



Eigenvalues: axis 1 = 0.694; axis 2 = 0.518

Figure 5.5 shows that restricted range species are also not evenly distributed across the dry forest sites sampled in Huatulco. There is clustering of plots within sites, particularly those of sites 1, 5 and 8. Table 5.5 shows that on the evidence of Sørenson's index, the mean similarity between plots within sites is greater than that of plots between sites. The difference is highly significant (Mann Whitney U test, all within-site Sørenson's scores against all between site scores, $Z = -10.193$, $p < 0.000$). It therefore appears that single site sampling is equally poor for capturing bioquality at a locality as it is for capturing diversity.

Table 5.6. Mean within-site and between-site Sørenson indices of similarity for restricted range species using the 0.45 ha sampling protocol in eight dry forest sites in Huatulco Mexico. Within-site similarity measured as the mean index score of all pairs of plots within a site, between-site similarity measured as mean index score of all combination pairs of plots from different sites.

	Site								
Index	1	2	3	4	5	6	7	8	Between site
Mean species/ plot as % of site	24.4	32.3	18.6	24.4	30.7	18.1	37.1	17.5	
Sørenson's	0.26	0.26	0.17	0.24	0.42	0.32	0.22	0.37	0.19

The reduction of the sampling universe from all species to only restricted range species did not produce a consistent response in the similarity indices calculated. Tables 5.3, 5.4 and 5.6 show that for some sites similarity increased as a result of sampling only restricted range species whilst for others it decreased. However, ranking of sites by species richness and by number of restricted range species were significantly correlated (Spearman's rho, 0.976, $p < 0.01$).

Restricted range species were not the subject of analysis by Gillespie *et al.* (2000) or Trejo and Dirzo (2002) and hence direct comparison cannot be made here with those surveys. However, the ratio of the standard deviation to the mean of the number of restricted range species at each of the eight sites in Huatulco was 0.34, whilst that of all species is 0.13. There is, therefore, greater variability between sites within a locality when only the restricted range species are considered and therefore confident estimates of the number of restricted range species at a locality would be expected to require greater sampling intensity.

5.4 Discussion

These results show that there is significant local variation in the diversity of species and the diversity of restricted range species between tropical dry forest sites within a locality. They also show that this variation is likely to affect significantly prioritisation when regional scale comparisons are made by sampling single sites at various localities. For example, for the Mexican assessment of Trejo and Dirzo (2002), given the confidence intervals calculated here, any one of four localities might be the most diverse, and therefore be considered the most important for conservation. Similarly for the Central American localities of Gillespie *et al.* (2000) any three of the most diverse localities might be the most important for conservation.

Sampling the tree diversity of one (or very few) tropical dry forest sites in a locality is unlikely to estimate species diversity, or diversity of restricted range species, with sufficient accuracy to enable confident prioritisation amongst localities. The work of Trejo and Dirzo (2002) and Gillespie *et al.* (Gillespie *et al.* 2000) was used here for comparative purposes. It is acknowledged that these authors did not necessarily set out to prioritise sites for conservation. In the case of the former the main interest was in elucidating the relationship between precipitation and species diversity. The latter showed that Gentry's (1995) suggestion that Central American tropical dry forest diversity might be depauperate compared to other dry forest required re-consideration in the light of their more extensive Central American dataset. However, it is inevitable that studies that list places by diversity metrics will be interpreted as site prioritisations, indeed this has been encouraged by Hanson (2004) who argued that too often biodiversity inventory is seen as an end in itself and more application of such studies to conservation management is needed. It is therefore necessary to consider carefully the sampling protocols on which such surveys are based.

The questions raised here are essentially ones of the scalar structuring of diversity. The direct comparison of single site inventories is done largely without consideration of scale in that sites are compared across regions as if the localities from which they come are similar. One of the few fundamental truths of ecology is that species number increases with area (Hubbell 2001) but the rate of increase is likely to be different at different sites. A species area curve derived from sampling within a single site cannot therefore be extrapolated to estimate a species area curve for several sites. The sampling of multiple sites at a locality allows empirical construction of a locality's curve and a more accurate estimate of the species richness.

The locality surveyed here is probably far from unique in its between-site variability. Martínez-Yrízar *et al.* (2000) note that in the dry forests of the Sierra de Alamos and San Javier in north-west Mexico, dominance and density were highly variable in the former whilst in the latter floristic composition varied with elevation, soil composition, aspect and steepness. It is also likely that the same concerns are applicable to other forest types. For example, using a similar sampling methodology, Williams-Linera (2002) showed that diversity was highly variable within a Mexican montane forest locality. Using the Morisita-Horn index, which measures similarity in species and their relative abundance and varies between 1 for complete similarity to 0 for complete dissimilarity, she showed that similarity between forests sampled varied between 0.8 and 0, with a mean of 0.239 ± 0.11, that is, a confidence interval that is 44% of the mean. Tropical dry forest in Mesoamerica is characterised by particularly high levels of disturbance and fragmentation (Murphy & Lugo 1995; Trejo & Dirzo 2000). Whilst the precise effects of disturbance on tree diversity in this biome are not well understood, it has been shown by Hamer *et al.* (2000) that disturbance can have highly variable effects on Lepidopteran diversity in Indonesian moist forests depending upon the scale at which that disturbance is measured. Such effects can only be accounted for in diversity assessments if variation is sampled at more than one scale, and such concerns should not simply be ignored in conservation prioritisation.

Recognising variability in species composition within localities also focuses attention on the choice of sites to be sampled. Often assessment exercises give little indication as to why or how a particular site at a locality was selected. There may be an argument for selecting sites that are likely to contain high diversity, as interest could be in conserving the best a locality has to offer and most localities are likely to have deforested areas of minimal diversity value that might be ignored. However, such decisions presuppose an understanding of the spatial structuring of biodiversity at a locality and thus anticipate the results of an assessment. It has been argued that various components of dry forest landscapes, including highly disturbed ones, might have potential for tree diversity conservation (Boshier *et al.* 2004; Gordon *et al.* 2004) and therefore would merit inclusion in an assessment. To dismiss what might appear to be unpromising sites in favour of one or a few mature forest sites risks missing important components of diversity. Only by sampling more than one site within a locality, and ideally through randomised selection of sites, can a full assessment of diversity at a locality be achieved.

As well as attention to the scalar structuring of diversity, it is suggested that greater attention in prioritisation exercises be given to threatened species, this being especially important in tropical dry forests which Gentry (1995) contends are at least as important for their high concentrations of endemics as they are for their species richness. It is inevitable that if the only assessment exercise available is one that does not consider threat status, it will still be used as a prioritisation guide. Yet with relative ease, information on restricted range species can be incorporated in prioritisation approaches, as illustrated here.

No general solution can be offered as to exactly how many sites should be sampled for a tree diversity assessment. This would be dependent on the forest type sampled, the pattern of spatial variation encountered and the precise objectives of an assessment. It is suggested that in rapid botanical surveying, sufficient randomly selected sites at each locality should be sampled to ensure that the means of the variables of interest become stable. For a prioritisation exercise, an insignificant alteration of the calculated mean would be one that does not change the ranking of a locality with respect to others. The variances of estimates of the mean must also be given consideration; the importance of a locality with several outstanding sites may be obscured if especially low scores from other sites reduce the overall mean. This may be particularly likely if a locality has undergone fragmentary disturbance-highly disturbed sites dominated by pioneers may disguise the few, but potentially important sites when a mean for the locality is calculated.

In conclusion it is proposed that the locations compared in regional assessments need to be sampled more intensively, that surveys should consider various landscape components at a locality and that some measure of species threat needs to be incorporated into assessments. All of this requires more resources in a discipline that is already severely stretched. However, this extra investment may be well rewarded if it leads to more efficiently targeted conservation actions. It was noted that the estimates of within location variation in composition, here measured by Sørenson's index, had narrower confidence intervals than estimates of species richness, suggesting that selection protocols that make use of the identities of species, such as complementarity, may be less demanding of sampling intensity. Fortunately more efficient inventory techniques are emerging that are rapid and robust in their application to varying landscapes (e.g. Sheil *et al.* 2003) and techniques for incorporating species threat into assessments being put into practice for various taxa (e.g. Dunn *et al.* 1999; Hawthorne 1996; ICBP 1992).

Chapter 6: Techniques for Efficient Floristic Inventory

6.1 Introduction

There is general agreement amongst biodiversity scientists that cost-effective conservation requires biodiversity assessment in order to guide the selection and prioritisation of sites for protection (Margules & Pressey 2000; Phillips *et al.* 2003; Pressey *et al.* 1993; Royal Society 2003). This is also recognised by international policy processes such as Agenda 21 and the Convention on Biological Diversity. At global and regional scales considerable progress has been made in setting conservation priorities for habitat types (Dinerstein *et al.* 1995) and for specific taxa including vascular plants and vertebrates (Ceballos *et al.* 1998; Davis *et al.* 1997; Heywood 1995; Myers *et al.* 2000). However, at local scales, that is within habitats regarded as priorities, there is less use of biodiversity assessment to guide prioritisation with the result that decisions about what to protect and where to protect it are driven primarily by socio-economic criteria. Such selection results in reserve networks that are inefficient as they inevitably contain fewer species than the theoretical maximum (Pressey 1994). Hence there remains a gulf between theoretical best practice and site selection as it usually occurs (Prendergast *et al.* 1999). This is especially so in lower income countries where a disproportionately high number of the world's most diverse terrestrial ecosystems are found (Myers *et al.* 2000) but where resources and the institutional capacity needed for biodiversity assessment are often absent or poorly developed. Particular interest is now focused on rapid biodiversity assessment (RBA) methods, as typified by Conservation International's Rapid Assessment Program (Abate 1992) which has developed and promoted efficient methods of characterising the diversity of particular regions through directly sampling species at sites of interest (e.g. Montambault & Missa 2002).

Biodiversity assessment is a broad term which has been used by different authors to include a number of different activities (e.g. Cantú *et al.* 2004; Dudley & Jeanrenaud 1998; Hilton-Taylor *et al.* 2000; Lawrence 2002; UNEP-WCMC 2003). Therefore, following Stork and Samways (1995), a distinction is made here between biodiversity inventory and biodiversity monitoring, both of which may be considered part of biodiversity assessment. Inventory and monitoring differ in that the former is a one-off process that identifies conservation priorities between species and habitats in a defined area at a given time. Inventory is typically used to identify the sites where investment in biodiversity conservation is likely to be most appropriate. In the general model of a conservation project proposed by Salafsky *et al.* (2002 p. 471) the objective of biodiversity inventory corresponds to defining 'the specific conservation target that the project ultimately would like to influence'. In contrast, monitoring is a dynamic process that aims to detect changes in the status of biodiversity and ideally should be put into practice only once important biodiversity has been identified through an inventory process. Thus, practices known collectively as RBA are usually inventory practices and the focus here is on protocols that are used in rapid inventory, specifically tree diversity inventory.

Biodiversity conservation in lower and mid-income countries is often funded by international and multilateral institutions, however, many international conservation organisations choose to work through national and local partner institutions (see chapters 2 and 8). Biodiversity inventory, if it is done well, can be demanding of both technical and financial resources and the acute financial and technical limitations often faced by such organisations mean that biodiversity inventory is likely to be passed over in favour of activities that are more immediately tractable. It is therefore necessary that a pragmatic approach to biodiversity inventory be adopted in which it is recognised that the highest standards set by biodiversity scientists are unlikely to be obtainable. A less ambitious but adequate inventory exercise is likely to result in considerable improvement in site selection efficiency over the alternative of no inventory input. Such a pragmatic approach is adopted here, with the limitations faced by small conservation organisations based in lower income countries very much in mind in selecting and comparing biodiversity inventory protocols.

Various field methodologies for the rapid characterisation of tree diversity have been proposed. Fixed area methods are the most widely used, with the commonest rapid assessment technique perhaps being the 0.1 ha method popularised by Gentry (1982) that uses ten repetitions of 2 x 50 m 'strip transects'. In one of the few attempts to directly measure and compare the efficiency of inventory methodologies, Phillips *et al.* (2003) tested the Gentry method against the use of 1 ha plots and showed that the former was more efficient as measured by Crude Inventory Efficiency (CIE), the number of species encountered against time searching. Furthermore, it was shown to be more statistically powerful because more repetitions per unit time were achieved. Hall (1991) made a case for methods based on a fixed tree counts, in which sampling effort is controlled by selecting a fixed number of trees per inventory, that do not require the demarcation of plots. He used such a method to successfully survey montane forest in Africa, however, he provided no direct comparison of its efficiency with other methods. Condit *et al.* (1998) also supported such fixed tree count methods on statistical grounds. They argued that by comparing equal numbers of stems the resulting diversity indices were not prone to biases resulting from differences in density that affect fixed area methods. Stern (1998) tested fixed area plots, the strip transects of Gentry (1982), against fixed tree counts, which she referred to as variable area transects, within completely inventoried 1 ha plots in Peruvian Amazonia. Her conclusions were that the fixed count plots were more flexible, particularly when different vegetation structures were encountered, but that strip transects had the advantage of being comparable to assessments from many other sites worldwide. However, she went on to question whether estimating species diversity is in itself worth the considerable investment it requires and suggested that a simple checklist of important or uncommon species, along with a brief structural description, might be a more useful conservation tool.

Variations on this checklisting or *ad hoc* approach are less well tested but are widely used, for example in Conservation International's Rapid Assessment Program (e.g. Schulenberg *et al.* 1999). With *ad hoc* methods, the control of effort put into sampling (which in other methodologies is done by fixing area or the number of trees) is minimal. Because of this lack of control, comparisons between the resulting checklists are difficult, and the lack of repeated sampling at each site prevents the calculation of confidence limits. Furthermore, checklisting approaches do not easily allow accurate estimation of abundances of the species encountered. However their advantages, principally of economy, have been highlighted by Droede *et al.* (1998) with respect to the monitoring of plants and animals, and they have been applied to the assessment of tree diversity by Gordon *et al.* (2004) and Hawthorne (1995).

The prioritisation of sites for conservation by species richness (e.g. Kerr *et al.* 2000), or even number of threatened species (e.g. Gordon *et al.* 2004) is a relatively crude means of increasing the efficiency of conservation networks. The concept of complementary reserve selection is now well established, at least in the literature (Gaston & Rodrigues 2003; Margules *et al.* 1988; Pressey *et al.* 1993), as a means of ensuring reserve networks contain all species locally recorded. Complementarity analyses typically employ one of a growing number of heuristic algorithms to prioritise sites by the contribution each can make to the total number of species within a reserve network. Hence a site's importance is not reflected in the number of species that it contains, but in the number of species that it contains that are not already represented elsewhere. Whilst much attention has been focused on the development of increasingly sophisticated algorithms (e.g. Briers 2001; Rodrigues & Gaston 2002) less emphasis has been given to the sensitivity of their outputs to variation in input data, as provided by different inventory approaches.

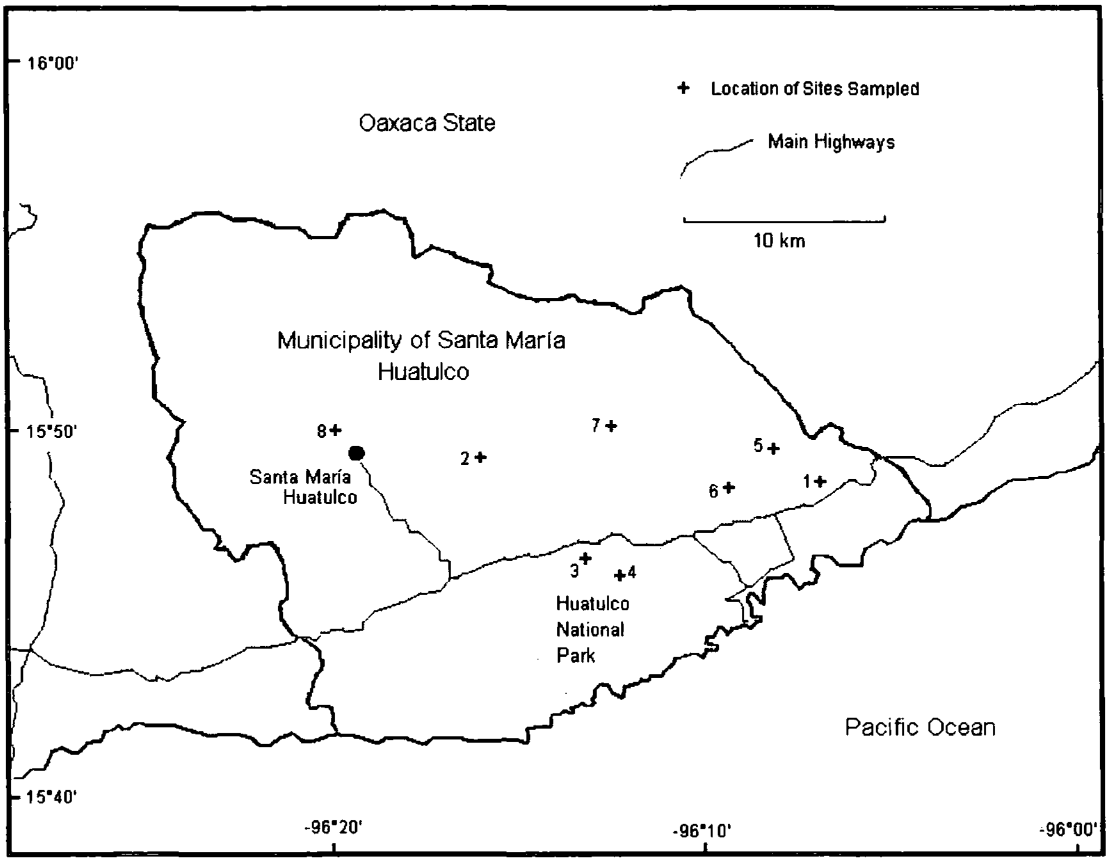
Whilst a number of tests of biodiversity inventory protocols have been carried out in tropical moist forest, such as those by Phillips *et al.* (2003) and Stern (1998), no such analyses have previously been performed in tropical dry forest. The aim of this chapter is to compare fixed area, fixed count and *ad hoc* methods for the rapid inventory of tropical forest tree and shrub diversity in eight seasonally dry tropical forests sites in southern Mexico. Here the sole objective of rapid inventory is assumed to be the identification of priority sites for conservation. Phillips *et al.* (2003) showed the 1 ha method to be inefficient for rapid inventory and is not tested but an *ad hoc* checklisting method is included. Efficiency is here defined following Phillips *et al.* (ibid) as the ratio of number of species encountered to the working time spent carrying out the inventory. Using this measure the efficiency each method is estimated and compared with further consideration given to the statistical robustness. The results of the prioritisation of sites, by species number and by complementarity, are compared under the assumptions that the target of conservation should be: a) to conserve maximum species richness and b) to conserve the maximum number of threatened species. Finally, recommendations are given for tree diversity assessment under conditions of limited technical and financial resources.

6.2 Methodology

6.2.1 Study Site

Eight areas of seasonally dry tropical forest were inventoried in 2002 in the municipality of Santa María Huatulco in Oaxaca, Southern Mexico (Figure 6.1). This area, on the Pacific lowlands and foothills of southern Mexico, contains a mosaic of agricultural lands and seasonal forests in various stages of succession. The tree and shrub diversity in this area is similar at familial and generic levels to other Mesoamerican dry forests (Gordon *et al.* 2004; Salas-Morales *et al.* 2003). The sites fall within an area of approximately 250 km^2 with an altitudinal variation of between 20 and 150 m.a.s.l.

Figure 6.1 Location of eight dry forest sites sampled in Santa María Huatulco, Mexico.



From the forest patches in the area of interest eight sites were selected randomly with the precondition that each should contain closed canopy forest. Some of the sites were undoubtedly mid-succession secondary forest with the ages of the youngest estimated, by conversation with long-term residents, to be between 15 and 20 years. Other sites may have been primary, although none could confidently be said to be unaffected by anthropogenic disturbance as the region has a history of commercial logging (Gordon *et al.* 2005) and extensive use by local farmers for extraction of forest products and livestock forage. The forest patches sampled varied considerably in size from under 50 ha to over 300 ha. Sampling within the patches was carried out at least 100 m from the forest edge and riverine forest was excluded.

One forest, site 8, was determined by visual inspection and prior to surveying to be the most floristically distinct of the sites, being atypically semideciduous and dominated by two species, *Astronium graveolens* (Anacardiaceae) and *Guarea excelsa* (Meliaceae) that were absent from, or very rare in, the other sites.

6.2.2 Protocols

Four inventory protocols were tested and timed, namely, two fixed area methods, one fixed count method and an *ad hoc* method.

Fixed area methods

The first fixed area method adopted was a variant of the method popularised by Gentry (1982) in which ten repetitions of 2 x 50 m strip plots are used. Here plots were randomly placed and for comparative purposes, the number of plots was increased to 15. This methodology is henceforth called the 2 x 50 m protocol. The practical advantage of narrow plots is that they require only a line of 50 m to be cleared in the forest from which it can be easily determined which stems fall within 1 m measured perpendicular to either side of the line, thus there is no need to circumscribe the entire perimeter of the plot. The 2 m width is, however, arbitrary and the same advantage can be derived from wider plots. Hence the second of the four methods is also a fixed area method in which effect of altering the size of Gentry's plots was tested by increasing the width of each to 6 m (3 m either side of the line). The result is here called the 6 x 50 m protocol.

Fixed count method

Fixed count plots require no plot demarcation and therefore are potentially less time-consuming over fixed area methods. Following the recommendations of Condit (1998) that fixed count methods should have a minimum of 100 stems per sample and of Hall (1991), that 15 tree plots should be used, a protocol was devised in which each plot comprised 15 trees closest to a central point. In effect, these are circular, variable-area plots, in which only the distance of the trees closest to the perimeter from the central point need be measured in order to determine which trees fall within the plot. Fifteen repetitions of this plot type were performed at each site and the methodology is here called the fixed count protocol.

For fixed area and fixed count methods, plots where randomly placed without replacement at each site. The time to establish each and measured to estimate the efficiency of each inventory protocol.

Ad hoc method

The *ad hoc* method was kept deliberately informal, it amounting to no more than a checklisting exercise in which the assessment team surveyed the area of forest sampled, starting at its perimeter and circling inwards until the team decided, subjectively, that no more new species were likely to be found. To assist with the comparative analysis the time at which each species was encountered and entered into the checklist was recorded.

General

The inventory of a site started with the *ad hoc* protocol, the execution of which determined the area in which the randomly placed plots of the other methodologies would be placed. This ensured that each protocol sampled the same area of forest at each site. During the inventories voucher specimens of tree species not identifiable in the field were taken for later identification at the national Herbarium of Mexico (MEXU). Voucher specimens were deposited at the QUIE Community Herbarium in Santa María de Huatulco. The threat status of each species was determined as in Chapter 5, with the 20% of species found (all samples combined) determined to be threatened. The field team was comprised of a data recorder, a tree spotter/climber and a labourer and was used throughout the survey. All three members of the team had prior knowledge of the local tree flora and therefore contributed to species identification. Because of the low stature of the forests collection of vouchers rarely required tree climbing and this therefore had an insignificant effect on efficiency.

6.2.3 Analysis

The number of species found for each site using each protocol was calculated and expressed as a percentage of the total species found at each site for all methods. Differences between the mean species number per sub-unit (2 x 50 m plot, 6 x 50 m plot and fixed count plot) were tested using the t-test. The similarity between each pair of sites for the four protocols was compared using the Jaccard index. The Jaccard index (Sj) varies between 0 and 1 where 0 is complete dissimilarity and 1 implies complete similarity, and is given by:

$$S_j = a/(a+b+c),$$

where Sj = Jaccard similarity coefficient, a = number of species shared by both samples, b = number of species unique to the first sample, and c = number of elements unique to the second sample.

The efficiency of the subunits used in the protocols was compared using species accumulation curves, with effort being measured first by the time spent by the inventory team carrying out the surveys and then by the number of individual trees encountered. The relative efficiency of each of the seven protocols was further compared by calculating inventory efficiency in a manner similar to the crude inventory efficiency of Phillips *et al.* (2003). Here efficiency is defined as the total time spent by a three-person inventory team in the field setting out and enumerating plots, measured in units of 'team-minutes', which is then divided by the number of species found in the sample to provide a measure of inventory effort per species. In comparing these efficiencies it was assumed that all other activities related to the assessments, such as time spent travelling to each site, were independent of the assessment protocol used. Whilst the number of species per sample is a very simple metric, it is perhaps the single most commonly used estimator of biodiversity. Here it was further refined by limiting the analysis to the number of threatened species per sample to investigate the relative efficiencies of inventories of all species and threatened species only.

The number of species found by sampling is always an underestimate of the total number of species in a forest, hence the higher the 'species observed' count that an inventory protocol yields, the more likely it is to guide to optimal conservation strategies. The ability of the protocols to confidently distinguish between observed species number at different sites was tested by comparing the mean number of species per plot after 15 repetitions, and for selected sites running means were plotted with confidence intervals to demonstrate how quickly different protocols distinguish between sites. Because of the lack of within-site repetitions, the *ad hoc* method was excluded from this part of the analysis. The total number of species is estimated for each protocol and each site using two non-parametric estimators, the incidence coverage estimator, (ICE) and Chao 2. In a comparative study of several such estimators Chazdon *et al.* (1998) showed that these two best satisfied the requirement of a species-richness estimator for a Costa Rican humid tropical forest. Both ICE and Chao 2 have the advantage of requiring only presence/absence data and use information on the occurrence of species that are found in only one sampling unit ('uniques') to calculate the number of additional species that are likely to be found outside of the fraction of forest that fell into the sample. Chao 2 has the additional advantage that a standard deviation can be calculated, and hence a confidence interval for the estimate of total species (Colwell 2004a). Formulas for ICE and Chao 2 are given by Chazdon *et al.* (1998).

The complementarity principle of reserve selection implies not a comparison of species number, or numbers of rare species, but a comparison of the identities of the species that make up the samples from each site. The Jaccard index was used to compare protocols for their ability to distinguish between sites, and the significance of differences in the pair-wise comparisons of the protocols is tested by the Mann-Whitney test of ranks.

To further test the performance of the sampling protocols with respect to complementarity reserve selection, sites were then ranked using the 'greedy' heuristic selection algorithm of Briers (2001). This algorithm was chosen because of its simplicity and long history (Margules *et al.* 1988) and also because many other heuristic algorithms for site prioritization that are now in use are variants of this basic algorithm (Sarkar forthcoming). It selects the most speciose site first, then the site that contains most species not already represented in the first sample, followed by the site with most species not already represented in the first two sites, and so on until all species are represented. The sites were ranked in the order that the algorithm selected them on the principal that sites selected first contribute more new species than sites selected later and are therefore more important to an optimal reserve design. The site rankings were then tested by Spearman's *rho*.

6.3 Results

6.3.1 Summary Statistics for All Species

Table 6.1. Summary statistics for tree inventories of eight tropical dry forest sites in Oaxaca, Mexico using four inventory protocols. S(obs) = species observed in the given protocol, Total S(obs) = number of species observed in the four protocols.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Mean	All sites
Total S(obs)	71	61	80	61	67	96	88	50	71.8	169
2x50 protocol:										
S(obs)	48	46	47	38	48	49	44	21	42.6	123
% of total S(obs)	67.6	75.4	58.8	62.3	71.6	51.0	50.0	42.0	59.8	72.8
6x50 protocol:										
S(obs)	58	54	67	48	56	74	67	30	56.8	141
% of total S(obs)	81.7	88.5	83.8	78.7	83.6	77.1	76.1	60.0	78.7	83.4
Fixed count:										
S(obs)	50	47	56	37	51	56	49	26	46.5	121
% of total S(obs)	70.1	77.0	70.0	60.7	76.1	58.3	55.7	52.0	65.0	71.6
Ad hoc:										
S(obs)	43	34	61	50	50	76	66	44	53.0	143
% of total S(obs)	60.6	55.7	76.3	82.0	74.6	79.2	75.0	88.0	73.9	84.6

Table 6.2. Jaccard coefficients of similarity between eight tropical dry forest sites in Oaxaca, Mexico based on total species found by all protocols for each site.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Site 2	0.49						
Site 3	0.47	0.51					
Site 4	0.48	0.46	0.57				
Site 5	0.50	0.47	0.48	0.45			
Site 6	0.49	0.5	0.56	0.43	0.54		
Site 7	0.43	0.46	0.54	0.54	0.46	0.53	
Site 8	0.15	0.15	0.18	0.16	0.11	0.16	0.17

Table 6.1 shows that site 8, the site determined by visual inspection to be most distinct was revealed by all protocols, except the *ad hoc* method, to be the least diverse as measured by species observed. Site 6 was revealed to be the most diverse. The *ad hoc* protocol and the 6 x 50 m protocol made the largest

contributions to the total observed species list for each site, with 84.6% and 83.4% respectively. Table 6.2 also confirmed the distinctness of site 8 in species composition. Its mean Jaccard coefficient of similarity with all other sites was 0.15 which was significantly different to the mean of 0.49 of all other between site comparisons (t-test: $p < 0.01$). The most speciose site, site 6, had a mean Jaccard coefficient of 0.49 (excluding comparisons with site 8) and therefore high diversity in this site was not the result of distinct species composition suggesting that the importance of this site will vary depending upon whether importance is measured by species number or by complementarity analysis. Whilst the *ad hoc* protocol revealed the greatest number of species, the 6 x 50 m protocol revealed a similar number, and because of the repetitions of sub-plots it is this latter protocol which is used to derive a pooled total species count and two pooled estimated species counts for the area as a whole. These are shown in figure 6.2.

Figure 6.2. Pooled species area curve and estimates of total species for eight Mexican dry forests using the 6 x 50 m protocol. S(obs) = Species observed, ICE = Incidence Coverage Estimator of total species number, Chao 2 = Chao 2 estimator of total species number.

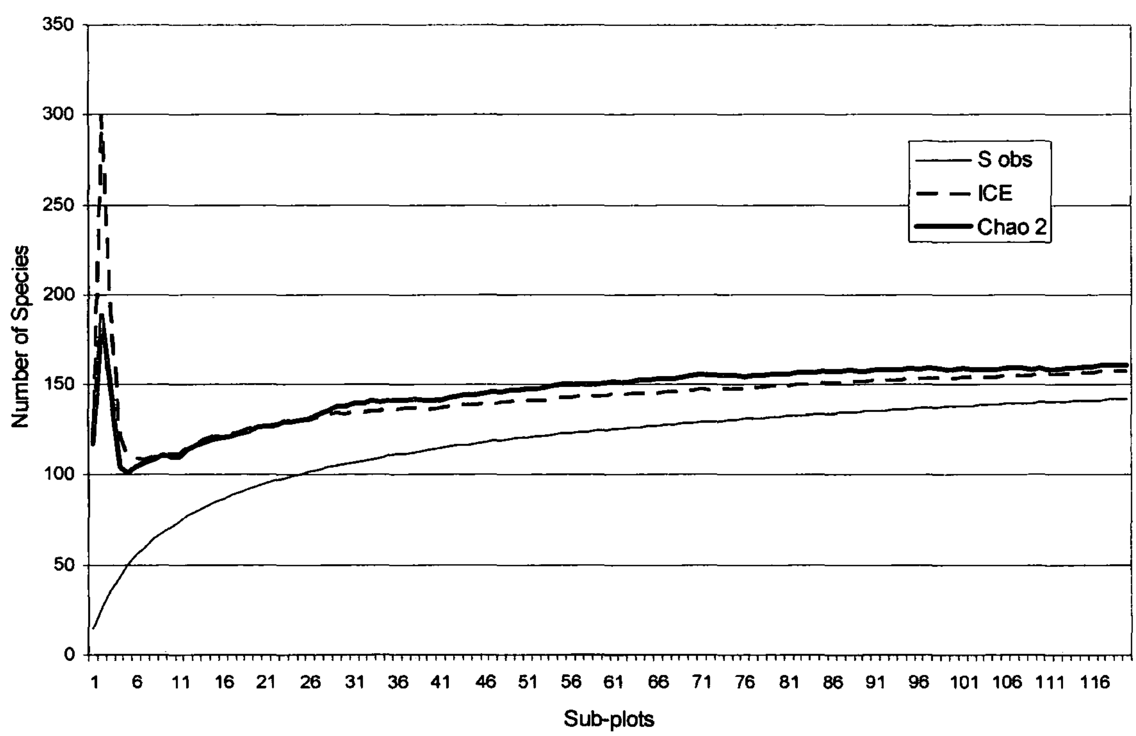


Figure 6.2 shows that the species observed curve, S(obs), which reaches a total of 141, is not yet at asymptote, and hence it is appropriate that estimators of total species richness are used (Colwell & Coddington 1994). Here Chao 2 estimates a

total species count of 161 and ICE estimates 157, suggesting that for this pooled sample, the observed number is between 87.6% and 89.8% of the true total. However neither of the two non-parametric estimators have yet to reach asymptote and therefore their estimates are not yet stable. This might be expected given that it is known that in these forests all sampling protocols combined revealed a total of 169 species (table 6.1).

6.3.2 Summary Statistics for Threatened Species

Table 6.3. Summary statistics for threatened species from eight tropical dry forest sites in Oaxaca, Mexico inventoried by four protocols. Total $TS_{(obs)}$ = number of threatened species observed by the four protocols, $TS_{(obs)}$= species observed in the given protocol. $TS_{(obs)}$ = number of threatened species observed.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Mean	All sites
Total TS(obs)	19	13	22	13	19	23	16	6	16.4	40
2x50 protocol:										
TS(obs)	14	10	11	7	12	12	12	5	10.4	28
% of total TS(obs)	74	77	50	54	63	52	75	83	66	70.5
6x50 protocol:										
TS(obs)	16	12	19	8	15	19	16	6	13.9	35
% of total TS(obs)	84	92	86	62	79	83	100	100	86	87.5
Fixed count:										
TS(obs)	14	9	15	7	13	15	11	5	11.1	31
% of total TS(obs)	74	69	68	54	68	65	69	83	75	77.5
Ad hoc:										
S(obs)	14	9	14	10	15	17	10	6	11.9	34
% of total TS(obs)	74	69	64	77	79	74	63	100	75	85.0

Table 6.4. Jaccard coefficients of similarity between content of threatened species in eight tropical dry forest sites in based on total species observed by all protocols for each site.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Site 2	0.22						
Site 3	0.42	0.35					
Site 4	0.44	0.19	0.60				
Site 5	0.33	0.33	0.53	0.25			
Site 6	0.36	0.37	0.41	0.29	0.33		
Site 7	0.28	0.27	0.26	0.25	0.25	0.42	
Site 8	0.12	0.15	0.05	0.07	0.05	0.10	0.14

Table 6.3 shows site 8 to be the least diverse as measured by threatened species observed. Site 6 was revealed to be the most diverse in terms of number of species threatened species. Note also that, as for total species, the *ad hoc* protocol and the 6 x 50 m protocol made the largest contributions to the total observed threatened species list for each site, with 85.0% and 87.5% respectively. Table 4 also confirms the distinctness of site 8 in threatened species composition. Its mean Jaccard coefficient of similarity with all other sites was 0.10 which is significantly different to the mean of 0.34 of all other between site comparisons (t-test: $p < 0.01$, distribution of Jaccard coefficients not significantly different from normal). These coefficients are reduced in comparison to their equivalents for total species; between-site similarity is reduced when comparisons are limited to the 20% of species of most restricted range.

6.3.3 Non-Parametric Estimation of Total Species Richness

Figures 6.3a, b and c show the species observed, $S_{(obs)}$, for each site for each of three protocols and nonparametric estimates of total species given by ICE and Chao 2. (Confidence limits cannot be calculated for ICE).

Figure 6.3a. Number of species observed and two nonparametric estimators of species richness for eight dry forest sites in Mexico using the 2 x 50 m protocol. S(obs) = Species observed, ICE = Incidence Coverage Estimator of total species number, Chao 2 = Chao 2 estimator of total species number.

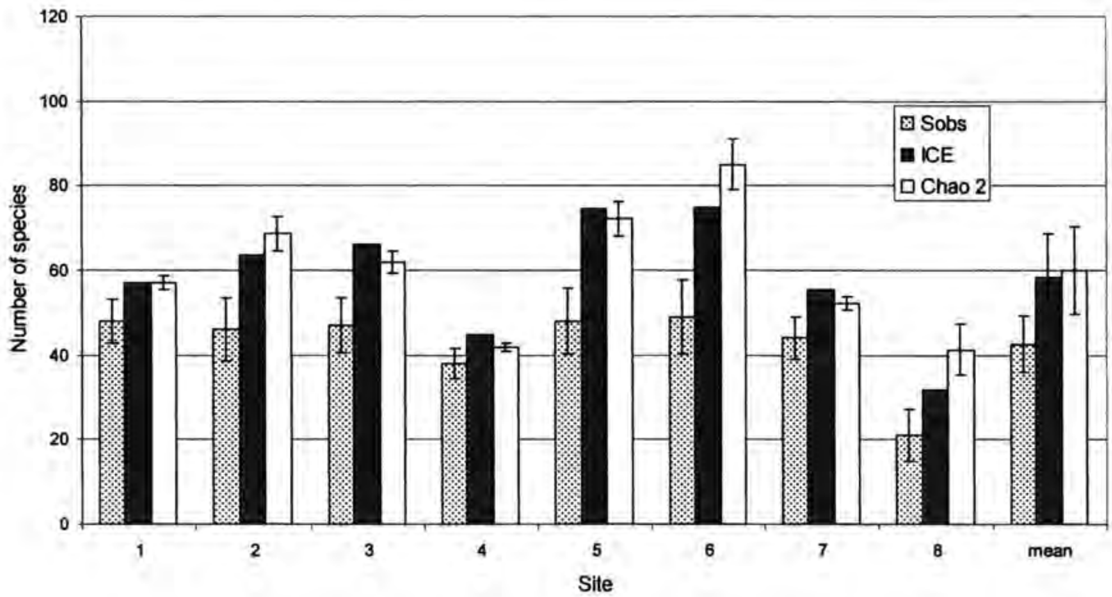


Figure 6.3b Number of species observed and two nonparametric estimators of species richness for eight dry forest sites in Mexico using the 6 x 50 m protocol. S(obs) = Species observed, ICE = Incidence Coverage Estimator of total species number, Chao 2 = Chao 2 estimator of total species number.

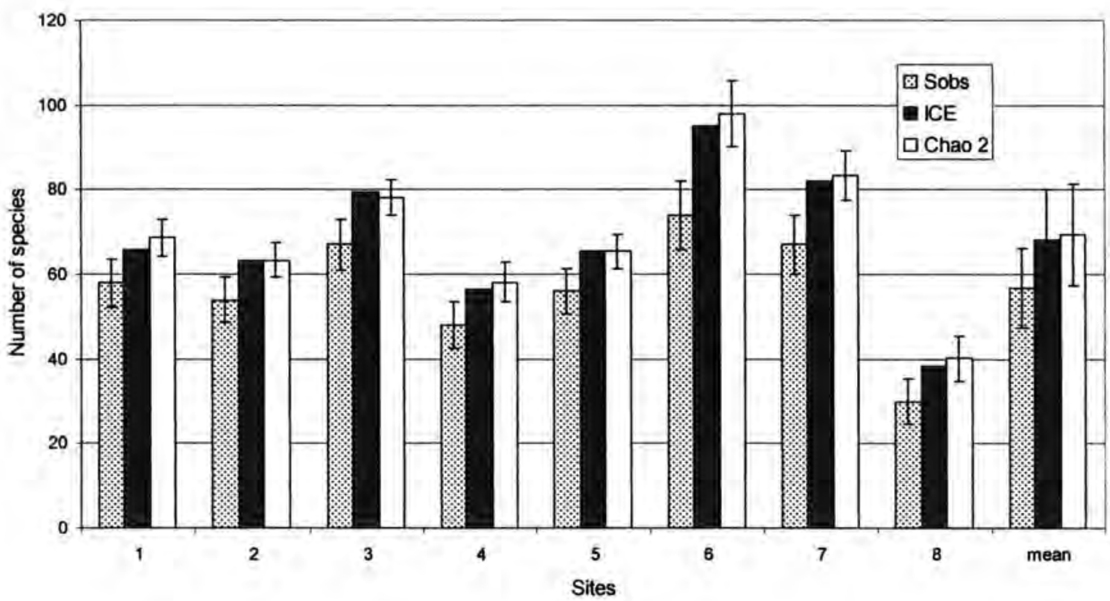
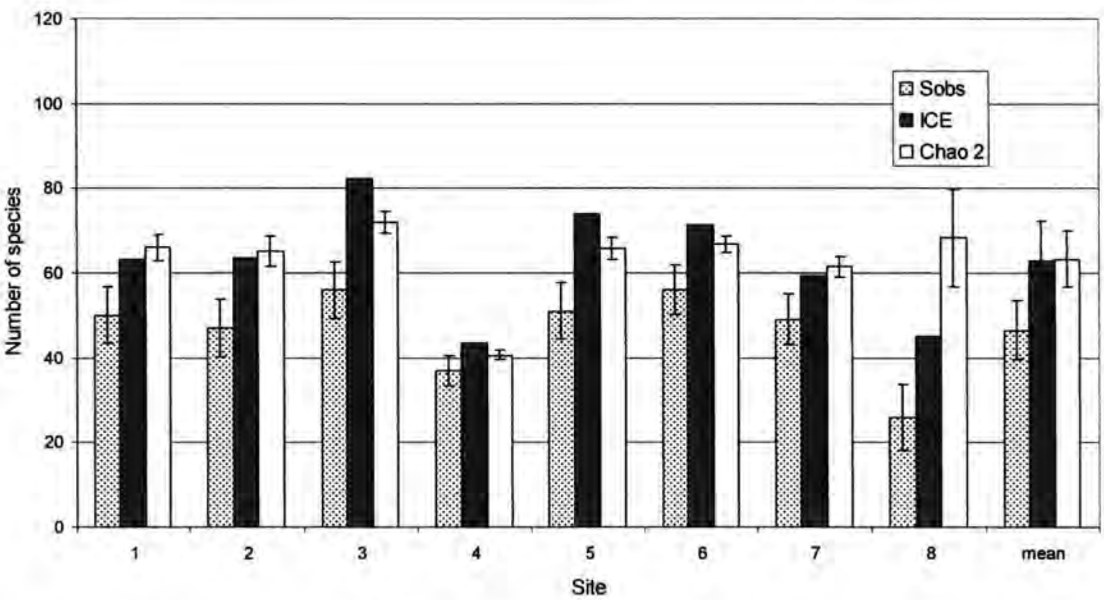


Figure 6.3c Number of species observed and two nonparametric estimators of species richness for eight dry forest sites in Mexico using the fixed count inventory protocol. S(obs) = Species observed, ICE = Incidence Coverage Estimator of total species number, Chao 2 = Chao 2 estimator of total species number.



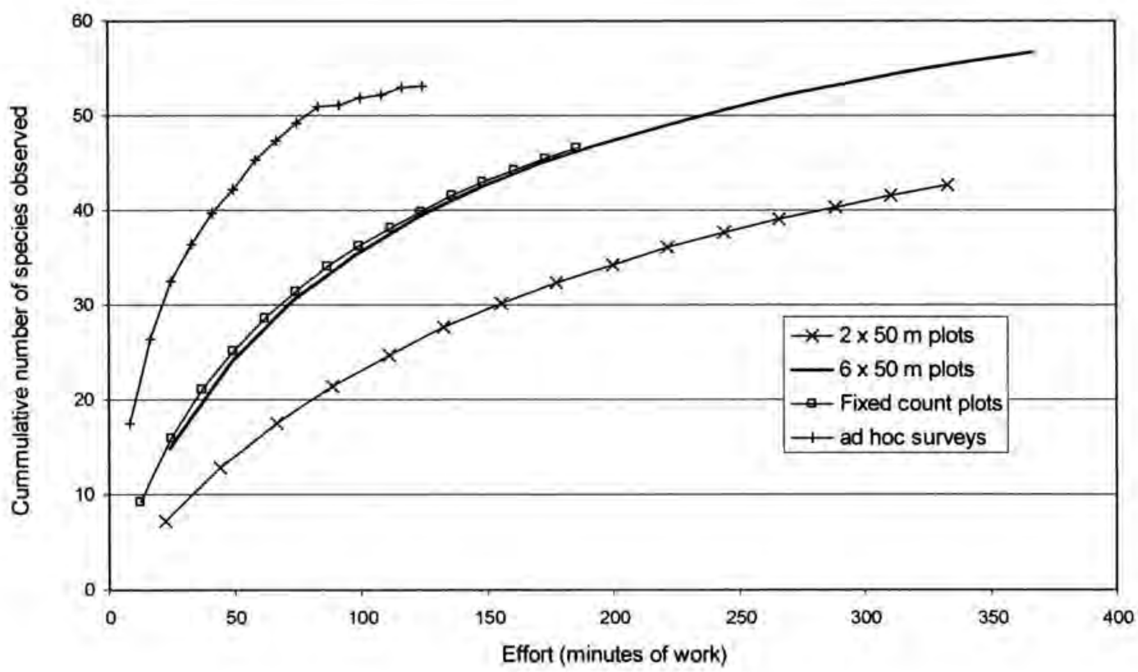
The Chao 2 estimator is shown to give a significantly different estimate of total species per site to S(obs) for seven of the sites sampled by the 2 x 50 m protocol, seven of the sites sample by the fixed count plots but only 3 of the sites when sampled by the 6 x 50 m protocol. Given that S(obs) of the 6 x 50 m protocol is closer to the total (but unknown) true species number, it is inevitable that estimates of total species are also closer to this value. Thus these non-parametric estimators are likely to be most useful where sampling fractions are lowest. Site 4 is consistent in not having a statistically different Chao 2 estimate to the S(obs). This suggests that this low diversity forest is relatively homogenous with very few species occuring in a single subplot, i.e. uniques, compared to the other low diversity site, site 8.

The estimate given by ICE falls within the 95% confidence limits of Chao 2, for all the sites when inventoried by the 6 x 50 m protocol, but only for a minority of sites for the 2 x 50 m and fixed tree protocols, suggesting that at larger sampling fraction ICE and Chao 2 estimates are similar.

6.3.4 Inventory Efficiency

Figure 6.4 shows the mean inventory efficiency across the eight sites of the four protocols measured by number of species encountered against the mean time the three-person inventory team spent on the protocol.

Figure 6.4. Efficiency of four inventory protocols in eight Mexican dry forests



In figure 6.4 the rate of species accumulation is measured by the gradient of the curve; the steeper the gradient the more efficient the protocol. All four curves start steep when 'new' species are rapidly encountered and then flatten off as additional species are encountered less often. This is the typical behaviour of the species accumulation curve in a diverse natural forest. However none of the curves is near to being asymptotic. By this measure the *ad hoc* protocol is the most efficient, its gradient is the steepest over the first 70 minutes of effort, or two thirds of its length. The abrupt flattening out of this curve after approximately 70 minutes of effort is an artefact of the inventory team being allowed to decide for themselves when the point had been reached at which no more species were likely to be found. The highlights one of the fundamental problems associated with *ad hoc* or checklisting methods of biodiversity assessment- the lack of control of effort.

The fixed count and the 6 x 50 m protocols trace similar curves and therefore have similar efficiencies. Thus by increasing the number of fixed count plots until the same amount of time had been spent on their enumeration as was spent on the 6 x 50 m plots, a similar $S_{(obs)}$ would be attained. It appears that the extra establishment time required for the 6 x 50 m plot is compensated for by the plots containing more individual trees (a mean of 27.4 per 6 x 50 m plot compared to 15 in the fixed count plot). However, the greater number of repetitions of sub-plots that would result from extra investment in the fixed count protocol would likely result in more precise diversity estimates of species number, that is, it is a more statistically efficient protocol. The 2 x 50 m protocol, the most widely applied protocol of those tested here, is shown to be the least efficient. Its high establishment cost is not compensated by a high number of individual trees per plot. On average only 9.1 trees were found in each plot.

6.3.5 Distinguishing Between Sites

The usefulness of a protocol lies not just in the number of species that it estimates that are present at a site, but in its ability to distinguish with statistical confidence between estimates of species richness from various sites. Tables 6.5a, b and c indicate that the mean number of species per sub-plot for three of the protocols are significantly different. Because the *ad hoc* method did not use repetitions it is not analysed here. The distribution of the means did not significantly vary from normality (Kolmogorov-Smirnov test, all $p > 0.05$) and hence the T-test could be used.

Table 6.5a 2 x 50 protocol: T-tests probabilities for pair-wise comparison of mean number of species per sub-plot for eight Mexican dry forests

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Site 2	0.74						
Site 3	0.53	0.73					
Site 4	0.48	0.28	0.19				
Site 5	0.76	1.00	0.75	0.31			
Site 6	0.70	0.43	0.30	0.72	0.48		
Site 7	0.55	0.77	0.93	0.18	0.79	0.30	
Site 8	0.01**	0.00**	0.00**	0.09	0.00**	0.02*	0.00**

* $p < 0.05$, ** $p < 0.01$

Table 6.5b 6 x 50 protocol: T-tests probabilities for pair-wise comparison of mean number of species per sub-plot for eight Mexican dry forests

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Site 2	0.51						
Site 3	0.15	0.45					
Site 4	0.12	0.04	0.01				
Site 5	0.96	0.55	0.17	0.12			
Site 6	0.32	0.72	0.69	0.02*	0.36		
Site 7	0.39	0.87	0.52	0.02	0.43	0.84	
Site 8	0.11	0.04*	0.01**	0.74	0.11	0.02*	0.03*

* $p < 0.05$, ** $p < 0.01$

Table 6.5c Fixed count protocol: T-tests probabilities for pair-wise comparison of mean number of species per sub-plot for eight Mexican dry forests

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Site 2	0.06						
Site 3	0.34	0.33					
Site 4	0.08	0.00**	0.02*				
Site 5	0.47	0.02	0.12	0.29			
Site 6	0.04*	0.91	0.28	0.00**	0.01**		
Site 7	0.68	0.22	0.69	0.07	0.33	0.18	
Site 8	0.00**	0.00**	0.00**	0.01**	0.00**	0.00**	0.00**

* $p < 0.05$, ** $p < 0.01$

Site 8, which was shown to be the most distinct of the sites in tables 6.1 and 6.2 is the only site which the three protocols could consistently distinguish, with statistical significance, by species number from the other sites. Site 6, the most speciose site, was shown to be significantly different by the 15 tree protocol to three other sites and by the 6 x 50 m protocol to one other site. Overall, however, the protocols did not perform well in distinguishing between these sites of similar habitat type. Recall also that for each of the protocols 15 repetitions of the sub-units were used, compared to the ten used by 2 x 50 m plots used by Gentry (1982). Fewer repetitions are likely to decrease the precision of estimates and thus make statistical separation of those estimates more difficult.

In figures 6.5a, b, and c the cumulative species curves of sites 4 and 6 are shown for each of the three protocols with 95% confidence intervals calculated by the analytical method of Colwell *et al.* (in press). These figures show that the confidence limits associated with the 2 x 50 m protocol were still overlapping after 15 repetitions although it appears that with a few more repetitions a statistical difference might be achieved. Conversely, for the 6 x 50 m and the fixed count protocol non-overlap of confidence limits is achieved rapidly, in both cases in less than 5 repetitions. This confirms the statistical superiority of the 6 x 50 m protocol and the fixed count protocol. However, it should be recalled that the effort required to establish and enumerate the 15 fixed count plots is far less than for the 15 fixed count plots, as the former require no plot demarcation, hence the fixed count plot curve in figure 6.4 is shorter.

Figure 6.5a Species accumulation curves for two Mexican dry forests sampled by the 2 x 50 m protocol.

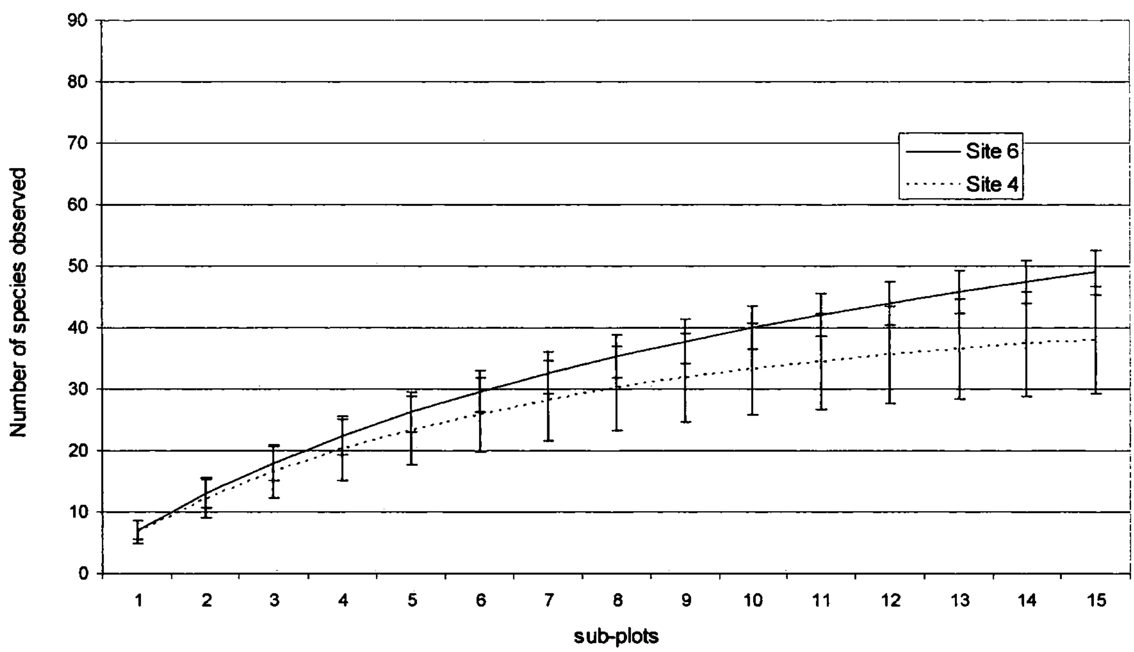


Figure 6.5b Species accumulation curves for two Mexican dry forests sampled by the 6 x 50 m protocol

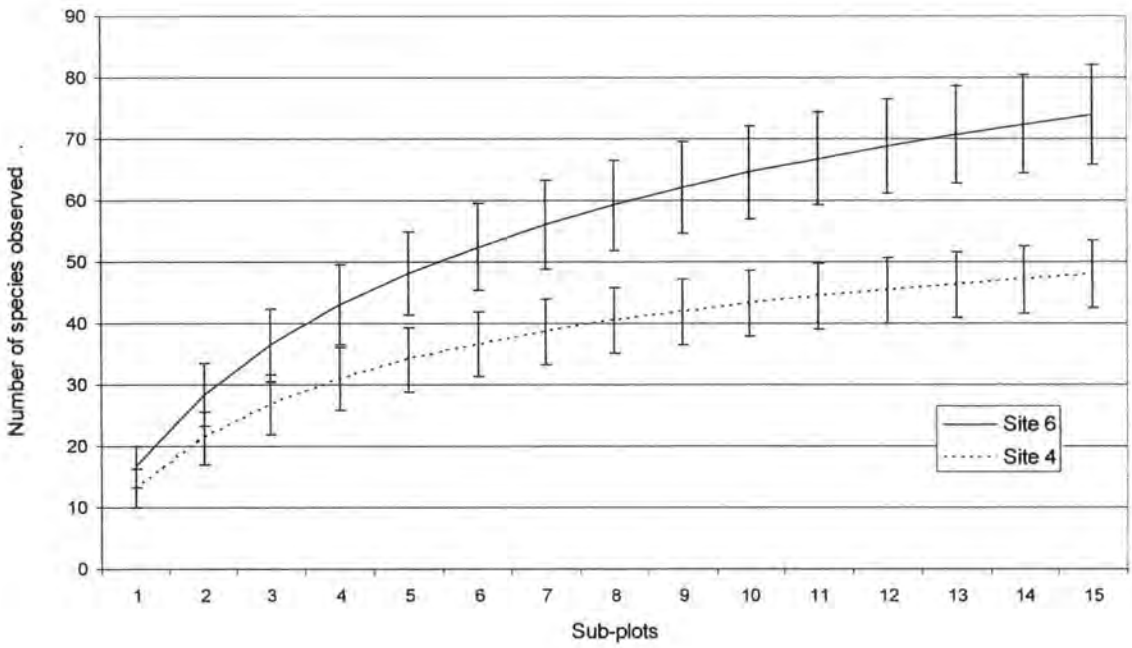
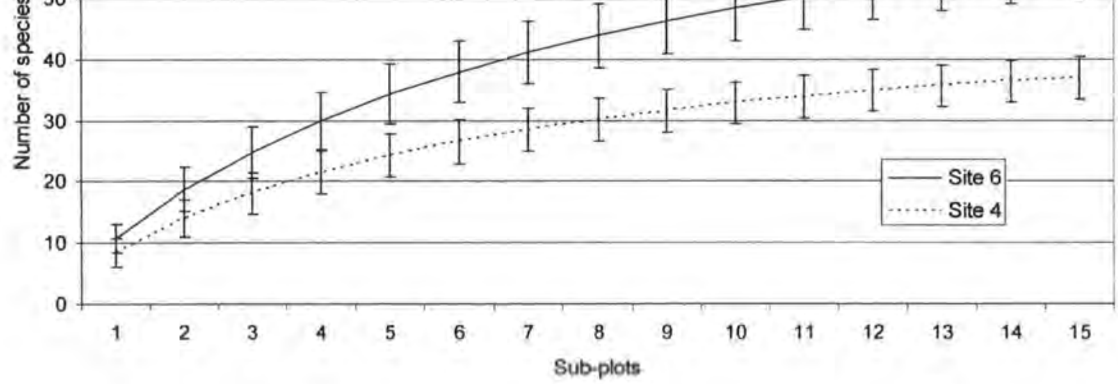


Figure 6.5c. Species accumulation curves for two Mexican dry forests sampled by the fixed count protocol



6.3.6 Performance of Inventory Protocols under Complementary Reserve Selection

The complementarity approach entails distinguishing between sites by the composition of species in each sample, rather than just the total number of species which has been the concern above. The differences in species composition between sites detected by each protocol is shown in tables 6.6a, b, c and d. Differences between sites are clear for all protocols, with few between site comparisons yielding similarities of greater than 50%. Mean similarities between sites are also similar, with the 6 x 50 m protocol detecting slightly larger differences.

Table 6.6a Between site differences measured by Jaccard coefficients for the 2 x 50 m protocol.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Site 2	0.40						
Site 3	0.48	0.43					
Site 4	0.39	0.35	0.47				
Site 5	0.55	0.42	0.38	0.34			
Site 6	0.43	0.38	0.30	0.30	0.33		
Site 7	0.35	0.48	0.36	0.32	0.33	0.37	
Site 8	0.06	0.05	0.06	0.04	0.08	0.04	0.07
						MEAN =	0.31

Table 6.6b Between site differences measured by Jaccard coefficients for the 6 x 50 m protocol.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Site 2	0.45						
Site 3	0.47	0.53					
Site 4	0.43	0.44	0.49				
Site 5	0.56	0.43	0.46	0.39			
Site 6	0.47	0.49	0.45	0.37	0.46		
Site 7	0.44	0.55	0.56	0.53	0.46	0.50	
Site 8	0.09	0.09	0.13	0.08	0.10	0.11	0.10
						MEAN=	0.38

Table 6.6c. Between site differences measured by Jaccard coefficients for the fixed count protocol.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Site 2	0.45						
Site 3	0.45	0.51					
Site 4	0.38	0.38	0.52				
Site 5	0.51	0.46	0.37	0.38			
Site 6	0.45	0.41	0.45	0.37	0.53		
Site 7	0.43	0.45	0.52	0.41	0.37	0.42	
Site 8	0.06	0.11	0.11	0.11	0.08	0.09	0.09
						MEAN=	0.35

Table 6.6d, Between site differences measured by Jaccard coefficients for the *ad hoc* protocol.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Site 2	0.35						
Site 3	0.41	0.38					
Site 4	0.43	0.40	0.56				
Site 5	0.39	0.38	0.39	0.39			
Site 6	0.38	0.33	0.52	0.40	0.40		
Site 7	0.33	0.33	0.43	0.45	0.35	0.42	
Site 8	0.14	0.16	0.17	0.15	0.08	0.14	0.16
						MEAN=	0.34

Differences between sites are clear for all protocols, with few-between site comparisons yielding similarities of greater than 50%. Mean similarities between sites are also similar, with the 6 x 50 m protocol detecting slightly larger differences.

Table 6.7 Mann Whitney U test for comparing ranks of Jaccard coefficients for four inventory protocols.

	2 x 50 m	6 x 50 m	Fixed count
6 x 50 m	0.00**		
Fixed count	0.04*	0.16	
Ad hoc	0.33	0.03*	0.29

* $p < 0.05$, ** $p < 0.01$

Table 6.7 shows that no protocol consistently resulted in Jaccard coefficients that were different to all the other protocols. No method can be said to be consistently better than the others are at detecting differences in species compositions.

However, the protocols did not always give a consistent order of priorities based on complementarity. Table 6.8 shows that no two protocols resulted in the same order of priority being selected by the 'greedy' algorithm, although some consistency is notable; site 1 was amongst the first four to be chosen regardless of protocol and site 4 was always amongst the lowest.

Table 6.8 Ranking of eight forests by five inventory protocols where rank is determined by the relative contribution of each site to reserve prioritisation by the 'greedy' selection algorithm.

	2 x 50 m (15 plots)	2 x 50 m (10 plots)	6 x 50 m	15 tree	*Ad hoc*
Site 1	7th	3rd	4th	5th	6th
Site 2	8th	8th	2nd	7th	7th
Site 3	2nd	7th	3rd	1st	8th
Site 4	6th	6th	7th	8th	5th
Site 5	5th	1st	6th	2nd	4th
Site 6	1st	5th	1st	4th	1st
Site 7	4th	2nd	5th	6th	3rd
Site 8	3rd	4th	3rd	3rd	2nd

Table 6.9 shows that correlations between the rank orders of sites are usually positive, however, similarities between the rank orders were found to be statistically insignificant. It therefore appears that, at least for this selection algorithm and forest type, complementarity analysis is highly sensitive to the dataset used. Tables 6.10 and 6.11 present a similar analysis, except that each sample is restricted to the 20% of species determined to be threatened. Again no two protocols resulted in identical rank orders, although sites 1 and 8 were consistently ranked amongst the highest, whilst sites 4 and 2 were consistently ranked amongst the lowest.

Table 6.9 Correlation of coefficients (Spearman's *rho*) of rankings of eight forests by five inventory protocols where rank is determined by the relative contribution of each site to reserve prioritisation by the 'greedy' selection algorithm

	2 x 50 m	2 x 50 m (10 plots)	6 x 50 m	15 tree
2 x 50 m (10 plots)	.071			
6 x 50 m	.347	.-419		
15 tree	.619	.238	.216	
Ad hoc	.500	.500	.096	.024

Table 6.10 Ranking of complement of threatened species of eight forests by five inventory protocols where rank is determined by the relative contribution of each site to reserve prioritisation by the 'greedy' selection algorithm. Sites not selected are those whose complement of threatened species are completely represented at other sites.

	2 x 50 m	2 x 50 m (10 plots)	6 x 50 m	15 tree	*Ad hoc*
Site 1	1st	1st	3rd	2nd	4th
Site 2	-	-	-	6th	7th
Site 3	5th	5th	1st	1st	-
Site 4	-	-	-	7th	5th
Site 5	6th	3rd	6th	-	2nd
Site 6	3rd	5th	2nd	4th	1st
Site 7	2nd	2nd	5th	5th	6th
Site 8	4th	4th	4th	3rd	3rd

Table 6.11 Correlation coefficients (Spearman's *rho*) of rankings of complement of threatened species of eight forests by five inventory protocols where rank is determined by the relative contribution of each site to reserve prioritisation by the 'greedy' selection algorithm. Sites not selected by the algorithm are ranked last (8th or 7.5th=) in this analysis.

	2 x 50 m	2 x 50 m (10 plots)	6 x 50 m	15 tree
2 x 50 m (10 plots)	.812*			
6 x 50 m	.639	.339		
15 tree	.587	.265	.850**	
Ad hoc	.263	.277	.096	-.238

* $p < 0.05$, ** $p < 0.01$

Compared to selecting sites by their relative number of species, which would result in the least speciose site, site 8 always being determined as the least important site, the greedy algorithm selected it as one of most important sites under all protocols, its importance being that it contains a high proportion of species not found at other sites. It is therefore probable that in selecting between more clearly differentiated sites, this algorithm would be less sensitive to the dataset used and hence produce more consistent rank orders.

6.4 Discussion

The test to which the four vegetation inventory protocols have been put here is a difficult one. Based on species number, $S_{(obs)}$, number of rare species and complementarity the aim was to distinguish between and prioritise eight forested sites, of which all but one are of essentially the same forest type. The similarity of these forests inevitably makes statistically significant separation of them difficult to obtain. However, this scenario is likely to face local conservation initiatives, and the organisations that undertake them. Given that no protocol successfully separated all the sites by species number with statistical significance even after fifteen repetitions, (either with all species or threatened species), the only conclusion that can be drawn is that the most efficient protocols should be used, but with sufficient repetitions to provide an appropriate level of statistical power, the precise number of repetitions needed obviously being specific to the forest type. An alternative, although perhaps less appealing approach, would be to carry out the maximum repetitions that resources allow of the most efficient protocol and then treat those sites that cannot be separated with statistical confidence as essentially the same forest type. What does emerge from this analysis is that the 2 x 50 m protocol with ten repetitions popularised by Gentry (1982) has little to recommend it in terms of efficiency or statistical power. Indeed its main advantage is that it is popular and therefore results from inventories carried out using this protocol are directly comparable to those from many other forests (Phillips & Miller 2002). This, however, is not likely to be the primary objective of a local prioritisation exercise. Indeed the need for such an exercise presupposes some form of existing assessment that determined the importance of the wider locality compared to others.

Conservation practice rarely follows an ideal trajectory and should the dual objectives of local prioritisation of sites and comparison of the forests of the locality of interest with those from elsewhere need to be combined, then the 6 x 50 m protocol may have much to offer. This protocol, with sufficient repetitions will more quickly distinguish between sites, but from each 6 x 50 m plot data a notional 2 x 50 m plot can be extracted providing the data recorder notes which individual trees fall within 1 m either side of the central 50 m line around which the plot is established. Thus better within-locality sampling can be achieved without compromising between-locality comparison. Otherwise the efficiency of fixed count circular plots gives them much to be recommended. This efficiency might further be enhanced by increasing the number of trees per plot, at least until the point where a significant number of trees per plot are obscured by other trees from the tree spotter standing at the centre of the plot. When this occurs the efficiency is likely to decline rapidly as measurements must be taken of the distance of trees at the perimeter from the plot centre to determine which trees fall in the plot.

The non-parametric indicators of total species richness consistently increased the estimates of species richness, however, for many of the sites the estimates were still below that of the total species observed calculated by combining each of the four inventories at each site (compare table 6.1 to figures 6.3a, b and c). This may be due to non-random distribution of species in each of these forests resulting from the clustering of species, a common phenomenon in tropical forests. Alternatively it may be because these estimators are not appropriate for estimates derived from relatively few repetitions in each sample; Chazdon *et al.* (1998) used over 80 replications of smaller plots to test a group of non-parametric indicators. They found that even the least biased estimate of species richness, the ICE estimator, did not approach a stable estimate (i.e. an asymptote) before 30 replicates were introduced. Whilst such richness estimators are useful in that they give an indication of how many more species are to be found at a site, they say nothing about the identities of those species. Thus such estimators, despite much current interest, are not useful for complementarity analysis.

A computational advantage of using complementarity for prioritisation is that the identities of species become relevant, hence the information gathered by each survey to be more effectively used. Two sites with a similar number of species are difficult to distinguish but may become easily distinguishable when species identities are considered. This was demonstrated here by the calculation of the Jaccard index that, for any two forests inventoried never rose above 0.56. This score was achieved for sites 1 v 5 using the 6 x 50 m protocol and for sites 3 v 4 using the *ad hoc* protocol (table 6.6b and 6.6d). The corresponding similarities in species number, expressed as ratios, were 0.96 and 0.82 (table 6.1).

Consideration needs to be given to the sensitivity of reserve selection procedures to data quality before it can be claimed that a prioritisation exercise provides an optimal solution to selecting reserve networks. The 6 x 50 m protocol gave the highest mean $S_{(obs)}$ for all the sites, yet as few as 60% of the total known species for site 8 were revealed by this protocol. There is no way of knowing whether these 'missing' species would be encountered elsewhere in a reserve network designed using these data, and therefore there is no way of knowing how efficient such a network would be. Given that species lists for tropical floras are invariable compiled by some form of sampling, controlled or *ad hoc*, this must be a concern for any reserve selection exercise.

The *ad hoc* inventory protocol showed considerable promise as a highly efficient methodology but also had severe draw backs. The tailing off of the *ad hoc* curve in figure 6.4 has a potential remedy, in that inventory teams could be made to search for similar lengths of time, however whether an inventory team, once convinced there was little more to find would search with the same commitment is contentious. Indeed the use of search time to measure effort is open to many uncontrollable anthropogenic influences. It is probable that an inventory team will search differently in forests of varying topographies, varying diversities or even, if fatigue becomes relevant, at different times of the day. Plot based methods which effectively force a team to input a similar amount of search effort are less susceptible to this. Given that control of search effort is critical to comparative inventory this has be a major concern. The *ad hoc* surveys analysed here were at least performed by the same team, whose knowledge of the floras of each forest was roughly similar. Yet in tables 6.8 and 6.10 these surveys were found to produce site rankings significantly different from the other protocols. Assessment exercises based on *ad hoc* 'checklisting' surveys often use different teams and no control of sampling time (e.g. Gentry 1995). The lack of repetitions typical of *ad hoc* or check-listing surveys also limits the statistical analysis to which they can be subjected, crucially making the calculation of confidence intervals impossible.

Finally, it can be said that the popular inventory protocol of ten repetitions of 2 x 50 m plots was shown to have underperformed compared to other protocols.

Chapter 7: Oaxaca's Network of Conservation Organisations

7.1 Introduction

In chapters 2 and 4 it was argued that biodiversity was a concept foreign to Oaxaca, and that it could not be argued that it is necessarily a better way of accounting for nature than are other more local conceptualisations, despite its prevalence in scientific discourses. Further, biodiversity assessment as currently practiced was shown to be less than ideal when measured against its own scientific merits, which suggests that those merits may currently be more rhetorical than epistemological. However, none of this necessarily leads to the conclusion that biodiversity, as a concept, cannot or should not play a role in the discourses and practices of nature conservation. Biodiversity is an actor that is already established amongst the institutions that mediate conservation and therefore has to be understood and dealt with in this context.

The aims of this chapter are primarily to describe the institutional relationships that mediate conservation in Oaxaca, southern Mexico, and to analyse those relations in terms of the formation of networks. In particular, it examines the relationship between local NGOs and transnational NGOs in the context of the role that scientific expertise has had in framing the institutional landscape of conservation in Oaxaca. It therefore describes important actors in terms of networks of interactions and translations. It then suggests reasons why biodiversity conservation does not necessarily follow the prescriptions for systematic planning and management that have been advocated by conservation scientists based in the higher income countries (the North).

The loss of biodiversity is seen by many as one of the gravest threats faced by humanity (Myers 1979) and an internationally coordinated response to this threat has been outlined in the Convention on Biological Diversity (CBD 2001). This response is given financial backing directly through associated funding mechanisms, (e.g. The Global Environment Facility - GEF) and indirectly through the various national and international organisations that have aligned themselves to the principles encapsulated by the CBD. Despite this, many biodiversity scientists based in, and speaking for, northern institutions lament the continued loss of biodiversity and the failure of many conservation initiatives (e.g. Heywood & Iriondo 2003; Salafsky *et al.* 2002; Terborgh 1999).

Raustiala (1997a; 1997b) argues that the people and institutions that have helped bring the CBD into being, and continue to support it, can be seen as an *epistemic community*. Haas (1992 p. 3) defines an epistemic community as:

> A network of professionals with recognized expertise and competence in a particular domain and an authoritative claim to policy-relevant knowledge within a domain or issue-area.

A feature of such communities is that they may be distanciated from the localities at which those 'issue-areas' are relevant. Indeed their interest in policy is precisely because they wish to extend their influence to such localities. Operating in global networks the epistemic community of the CBD has shaped a set of global policies for the conservation of biodiversity. Science and scientific practices are central to biodiversity conservation, from the formulation of the problem that has come to be known as the biodiversity crisis, to the solutions that are offered to that problem (Takacs 1996). The epistemic authority that supports the biodiversity agenda is therefore derived largely from sciences and scientists. Together, both problem and solutions are here referred to as the *biodiversity agenda* (see chapter 2). The modern representation of nature that is encapsulated in the concept of biodiversity has therefore become articulated and institutionalised as an actor (or 'actant') in global politics through its performance in scientific studies and discourses (Latour 1999).

Transnational conservation NGOs such as The World Wide Fund for Nature (WWF), The Nature Conservancy, Birdlife International and Conservation International are part of this epistemic community (Haeuber 1992; Raustiala 1997a; Redford *et al.* 2003; Sampford 2002). These transnational NGOs - also known as international conservation organisations, ICOs (Romero & Andrade 2003) - are an important conduit through which the biodiversity agenda is articulated and accordingly WWF is amongst the central actors considered in this chapter. They fund conservation science and employ conservation scientists (da Fonseca 2003) and are important as implementing agencies for conservation programmes and projects. Typically, transnational NGOs are based in, and funded from, higher income countries whilst much of the biodiversity they most wish to conserve is located in the tropics and sub-tropics of the mid and lower income countries (the South). Understanding conservation in lower income countries therefore requires an understanding of how scientific representations and transnational NGOs articulate and negotiate with institutions in the South. Increasingly that articulation is done via partner organisations, often NGOs, based in lower income countries. As Buergin (2003 p. 376) points out, nature conservation is now a global discourse of 'shifting power relations between local, national, and global levels of social organization'.

Whilst globalisation and transnationalism are less of a modern novelty to organisations in the South than it is often assumed (Bebbington & Batterbury 2001), in nature conservation the networking of the global and the local is currently especially relevant with the advent, in the 1990s, of international institutions such as the CBD and the GEF (Raustiala 1997b). Hence consideration of the power relations and negotiations between transnational NGOs and local NGOs is critical to our understanding of how the biodiversity agenda is enacted. This is especially relevant as a common perception amongst proponents of the biodiversity agenda is that conservation planning and implementation at the local level have failed to follow scientific prescriptions for identifying and managing conservation interventions (e.g. Flaspohler *et al.* 2000; Margules & Pressey 2000; Prendergast *et al.* 1999; Pressey *et al.* 1993; Salafsky *et al.* 2002). The result of this, they argue, is that conservation initiatives are typically inefficient and sub-optimal; less species are preserved than those scientific prescriptions would suggest should be possible given a certain level of investment.

In Oaxaca a manifestation of this problem is the lack of local biodiversity assessment to guide the location of protected areas. Whilst the scientific 'fact' that Oaxaca is biologically diverse has come to be accepted, the choice of areas to be conserved within Oaxaca has been largely determined locally by socio-economic criteria. Thus, Oaxaca provides a suitable case study for illuminating the wider 'problem' of the underperformance of international conservation initiatives. The question addressed in this chapter is, why is it that, despite the apparent authority and financial resources of the biodiversity agenda, biodiversity conservation has not translated wholly and unmodified to Oaxaca? What is under scrutiny is whether the intuitional relations that have come into being in Oaxaca are suitable for the enactment of the biodiversity agenda. This is essentially a question about how global concerns are enacted and contested at a local level, and, given that a community of non-governmental organisations is at the centre of this translation, answering it also reveals much about the relationship between global and local NGOs.

This question is addressed here through consideration of the degree to which an actor-network is successful in translating ideas and influences from diverse places and bringing them to bear on Oaxaca. Murdoch (1998 p. 362), suggests that actor-networks can be theorized as spaces that:

> [C]ome to be connected in ways which permit certain actors (or centres) to determine the shape of others, from a distance.

In his typology *networks of prescription* are those in which 'translations are perfectly accomplished', whilst *networks of negotiation* are those where:

> [T]he links between actors and intermediaries are provisional and divergent, where norms are hard to establish and standards are frequently compromised.

The success, or otherwise, of the extension of the biodiversity agenda into Oaxaca is appraised with respect to the relative predominance of prescription and of negotiation within its networks. This appraisal is based on an interview survey of senior staff in the principal organisations mediating Oaxacan conservation. Following a description of the methodology, the attraction of Oaxaca as a place to conserve biodiversity is described. This is followed by a description of the institutional landscape of Oaxacan conservation, starting with a consideration of the state and how it has allowed the non-governmental sector to take a prominent role in the implementation of conservation in Oaxaca. The non-governmental sector is then discussed, particularly the relationship between the group of Oaxacan conservation NGOs and two of their most important funding partners, The World Wide Fund for Nature and the Mexican Fund for Nature Conservation (FMCN by its Spanish acronym). The translation of the biodiversity agenda is then discussed in the light of the rural development agendas that are also pursued by Oaxaca's NGOs.

7.2 The Interview Survey and its Methodology

7.2.1 Selection of Interviewees

The organisations interviewed were chosen primarily through the author's previous knowledge of Oaxacan conservation; however, this was supplemented by consultation with the *Directorio Mexicano de la Conservación* (FMCN 2001). Relevant organisations included non-governmental, governmental and research institutions, the unifying characteristic being the involvement of each in biodiversity conservation, either through provision of funds, as regulatory authorities, as research institutions or as implementers of projects and programmes. Biodiversity conservation has multiple interfaces with education and rural development activities and hence many Oaxacan organisations can claim an interest in conservation, without necessarily being proactive actors in Oaxacan conservation. Such organisations, both governmental and non-governmental, are not the immediate focus here, but the opinions expressed by the staff of such organisations have also informed this chapter. The principal focus of this chapter is a group of Oaxacan based organisations whose principal institutional objectives include the conservation of biodiversity. Six locally based NGOs and one transnational NGO, WWF, active in Oaxaca form the core of the case study reported here. Brief details of those six local NGOs are given in table 7.1 They are referred to as conservation NGOs even though for some rural development may be an important part of their institutional objectives. Indeed the ways in which rural development and biodiversity conservation articulate is a recurrent theme in the practice of land management in the world's biodiversity hotspots (Brechin *et al.* 2002; Brown 2003; Marcus 2001; Wainwright & Wehrmeyer 1998). In obtaining an interview from at least one senior member of staff in each of these organisations it can be asserted that a large majority of the NGOs active in Oaxacan conservation are included in this case study.

The majority of interviews were carried out in Oaxaca, but a number of institutions relevant to Oaxacan conservation operate at a federal level and are located in the capital, Mexico City, where some of the interviews took place. In order to provide some perspective on the situation discussed in Oaxaca, a limited number of interviews were also carried out in the neighbouring state of Chiapas with representatives of local and international NGOs promoting biodiversity conservation there.

Table 7.1 Descriptive characteristics of the six locally based conservation NGOS discussed. All details are indicative rather than comprehensive.

NGO	Office location	Habitats of interest	Main funders since 2000	No. of technical staff as of 2000	Note
Naturaleza	Oaxaca City, with secondary office in dry forest	Dry forest, also shade coffee	Ford Foundation, WWF, FMCN	4	Approach biodiversity conservation as a tool for rural development. GIS capability.
Ecodesarrollo	Oaxaca City	Various	SEMARNAT, WWF, Consultancy	3	Approach biodiversity conservation as a tool for rural development. Strong interest in forest management.
Arbol Verde	Oaxaca City	Dry forest, shade coffee & montane forest	MacArthur, WWF, FMCN, CONACYT*	3	Strongly orientated towards research for biodiversity conservation. Advanced GIS capability.
Mesa Rural	Oaxaca City	Various	WWF, FMCN,	3	GIS analysis is principle input to community development and conservation.
Ecobosque	Oaxaca City	Montane forest	WWF, FMCN, World Bank	5	Community development as a means to biodiversity conservation.
Ecología	Oaxaca City secondary on coast now closed.	Various	MacArthur, CI**, FMCN, SEMARNAT,	5	Rural development & biodiversity cons-ervation within a framework of watershed management. GIS capability.

*CONACYT: Consejo Nacional de Ciencia y Tecnología, Government research council.

**CI: Conservation International

In the analysis that follows all interviewees' names are pseudonyms and, similarly, the locally based Oaxacan NGOs are referred to by pseudonyms. Other organisations have not had their names disguised as the context of the description and discussion makes their identity obvious.

The case study is informed by 20 interviews carried out in between May 2002 and June 2003. One, or in two cases two, senior members of staff were interviewed from each organisation. Where two staff members were interviewed, they were interviewed separately. Along with the six core local conservation NGOs, interviews were also obtained from members of staff of the National Herbarium in Mexico City, the FMCN, a senior manager of WWF's Oaxaca office and a representative of WWF-UK with responsibility for Mexico. Also interviewed were senior staff of *La Gringa* National Park and Huatulco National Park, The State Ecology Institute (IEE); the Ethnobotanic Garden of Oaxaca; an ex-manager of the Oaxacan office of The Ministry of the Environment and Natural Resources- (SEMARNAT) and a researcher at The National Commission for the Understanding and Use of Biodiversity (CONABIO). In Chiapas staff from two local NGOs, and from the regional offices of Conservation International and The Nature Conservancy, were interviewed about Chiapan conservation and their impressions of conservation in Oaxaca.

7.2.2 Interviews

Semi-structured qualitative interviewing was the principal research tool chosen for this part of the investigation. This investigation sought to elucidate the effects of a complex set of factors on the aims and activities of various types of organisation. This necessitated an open-ended approach to interviewing to accommodate questions and answers unique to each institution. Some basic topics and questions were, however, relevant to all institutions were raised in each interview, hence the use of semi-structured interviewing. Interviews were held in the interviewees preferred language, usually Spanish. The ethical considerations discussed in Mason (1996) were used to guide interview practice. Textual analysis was carried out using the software NVivo 2.0 (QSR-International 2002).

Interviewees directly quoted here are:

- Héctor, a biologist and director of *Naturaleza*
- Mario, the only Oaxacan interviewed here, works for *Mesa-Rural* .

- Yolanda, a biologist and director of *Arbol Verde*,
- Roberto leads *Ecodesarrollo*.
- Hugo an anthropologist and director of *Ecobosque*.
- Alejandro, a Oaxacan agronomist is director *of Campos Oaxaqueños,* an NGO not specifically concerned with biodiversity conservation, that works with indigenous communities on sustainable rural development and youth training.
- Gustavo, until a few weeks before being interviewed, was a senior manager in the Oaxacan office of SEMARNAT.
- Isabela is a long-standing manager in WWF's Oaxaca office and Sally is a programme director for WWF-UK stationed in the Mexico City office.

7.3 Oaxaca and its Biodiversity

The state of Oaxaca on the Pacific coast of southern Mexico is here used as a case study because its global importance as a biodiversity 'hotspot' has been established by the epistemic community of biodiversity conservationists. The state's mountainous topography and various climatic regimes have given rise to a remarkable diversity of habitats, ranging from high altitude semi-desert *matorral* to lowland tropical rainforest. It contains several priority *ecoregions* as defined by the World Bank/WWF Global 200 assessment of biodiversity (Dinerstein *et al.* 1995 – see below), an *endemic bird area* (ICBP 1992) and is part of the Mesoamerican hotspot of Myers (2000). In this sense Oaxaca can be described as a 'biogeographically imagined community' (Bryant 2002 p. 275) for biodiversity scientists. Because of this internationally recognised importance, it has attracted significant national and international funding for biodiversity conservation and the interest of local and international NGOs. One of the principal actors in Oaxacan conservation is the WWF whose regional office, in the state capital, Oaxaca City, is funded through the UK section of the WWF network. Oaxaca City, as the government and administrative centre of the state, is also the location of the main offices of most of the community of local NGOs involved in biodiversity conservation.

Despite such interest, biodiversity conservation in Oaxaca has not followed the prescriptions of biodiversity scientists, as is exemplified by selection of sites for conservation management in the dry forest zone in the lowlands along the Pacific coast of the state. Within this area the selection of reserves has been largely opportunistic, that is, informed by socio-economic feasibility rather than by biological criteria. Protected areas have emerged from land left over following expropriation for tourism development, and from communal areas that farmers consider either excess to their requirements for cropping or that they consider important for watershed management. Within the dry forest zone, no single site can be said to have been chosen because of its outstanding biodiversity value compared to other sites in the zone.

Few of the NGO staff interviewed doubted that it was a scientific fact that Oaxaca is especially diverse biologically, a belief that has been of considerable influence in them choosing to work in Oaxaca. This has taken on the form of received wisdom, no longer questioned perhaps because it is now in nobody's interest to do so, and nobody seems sure exactly where this fact is written, only that, as Héctor of *Naturaleza* suggested:

> It's already written, it's been said by many authors for several years, there are lots of things already published.

What is important is that everybody now agrees that Oaxaca is biologically very diverse. This 'fact' now seems to gather its strength as much from the way it articulates with the multiple actors in Oaxacan conservation as it does from the scientific methods and methodologies with which it was 'constructed'. The actors in Oaxacan conservation now need Oaxaca to be biodiverse to justify their interest in the state. However some, in this case Roberto of *Ecodesarrollo*, are occasionally prepared to question this apparently authoritative fact, noting the correlation between the amount of biological work done in a place and the number of species found:

> You'll know that Veracruz was the most biodiverse state because there were lots of specimens in the herbaria, because lots of collecting work had been done and because there were lots of publications, and then as people got tired of Veracruz they started to go to Chiapas and Chiapas became fashionable.

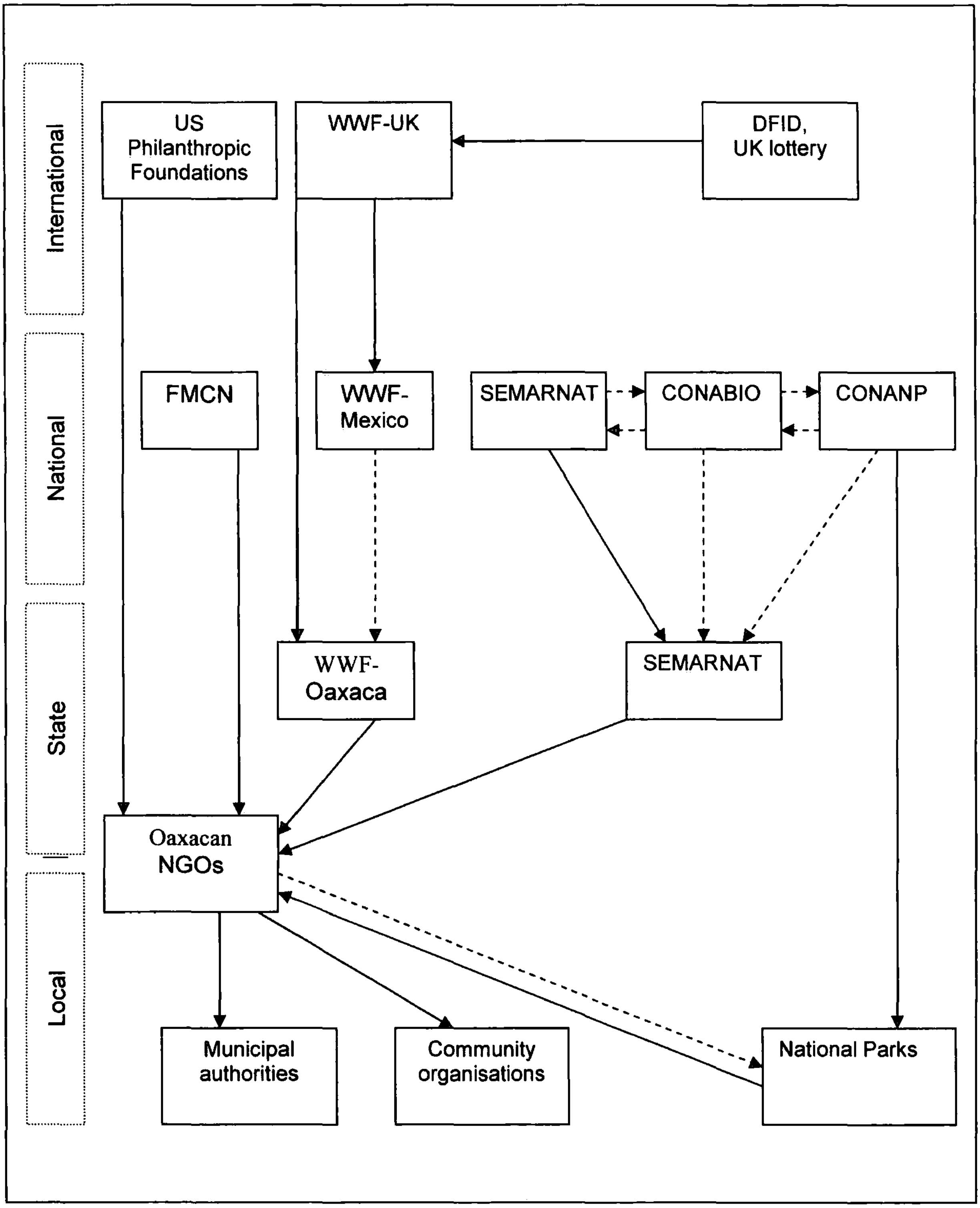
Notwithstanding this, the importance of Oaxacan biodiversity is now established globally as a scientific fact and hence the global politics of the biodiversity agenda are brought to bear on Oaxaca.

7.4 The Institutional Landscape of Oaxacan Conservation

7.4.1 Institutional Actors in Oaxacan conservation

The institutional relations discussed in this chapter are summarized in figure 7.1. This figure does not attempt to do justice to the complexity of institutional interactions relevant to Oaxacan conservation but is a schematic summary of the relations discussed in the chapter. Acronyms are expanded in the text on pages 12-13.

Figure 7.1 Simplified summary of the relationships between institutions described here. Solid lines are flows of information and finance, dashed lines are mainly information flows.



7.4.2 The State and Oaxacan Biodiversity Conservation

SEMARNAT

Despite a growing influence of non-governmental organisations, the state maintains an important role in nature conservation in Oaxaca, most particularly through SEMARNAT. SEMARNAT is a federal ministry, with separate delegations in each state capital, whose general remit is to protect the environment and promote sustainable development. SEMARNAT has a commitment to the concept of biodiversity. The word is used freely by its employees in Oaxaca; note however, that there is a clear understanding of its origins in scientific discourses and of its translation to other sectors/discourses, as Gustavo, previously of SEMARNAT Oaxaca explained:

> It is a new concept in our country, and it is used principally by specialists in the area of ecology and biology, however, in the area of economics it is also being used....and it is starting to be used in other sectors such as the rural sector because it is being shown and seen that biodiversity is a an important part of national heritage.

Under the current national government of Vincente Fox (2000-2006) there has been a redefining of the role of SEMARNAT. Previously its responsibilities covered regulation, research, promotion and implementation in the forestry, agriculture, water and fisheries sectors. As well as its fisheries remit being removed, it is now primarily concerned with co-ordination and regulation, with some responsibilities passed to other less centralised government bodies, such as the CONABIO and The National Commission for Protected Natural Areas (CONANP), which are further discussed below.

Much of SEMARNAT's implementation role is now fulfilled through out-contracting often, but not exclusively, to private profit, or not-for-profit organisations, the latter including Oaxaca's conservation NGOs. This is reflected in the Oaxacan office by a strong commitment to working with civil society, which here is taken to include the non-governmental sector, Gustavo, again:

> It's very important to have a good relation with civil society given that normally it's civil society that takes charge of the evaluations, the observations and commenting on policy.

Thus biodiversity conservation in Oaxaca has increasingly become the remit of local NGOs.

CONABIO

Despite its lack of a research remit, SEMARNAT can still access and even solicit scientific work related to biodiversity from CONABIO. CONABIO is an inter-ministerial federal commission based in Mexico City. It, like FMCN (see below), was constituted in 1992 as a response to the Rio Earth Summit. Its remit is to create and maintain a national system of biodiversity information, to support biodiversity conservation activities both by the state and non-state entities, and to support Mexico's commitments to international conventions related to biodiversity conservation, primary amongst these being the CBD. Its funds, which come principally but not exclusively from the federal government, are managed through a trust fund administered by a committee that includes representatives from governmental and non-governmental organisations. This partial detachment from government is another example of biodiversity conservation being devolved away from the state. CONABIO deals principally in scientific information and represents the most obvious commitment of the Mexican state to nature as *biodiversity*. It does not however have a direct role in the implementation of biodiversity conservation.

Protected Areas and CONANP

Protected areas are one of the most obvious manifestations of the politics of biodiversity conservation. Oaxaca's first two national parks, Lagunas de Chacahua and Benito Juárez were both established in 1937 (Simonian 1995). Decades of inactivity then followed, and it was not until the 1990s that Oaxaca's system of protected areas were expanded with the declaration of Huatulco National Park, and of the biosphere reserves of Cuicatlán-Tehuacán and La Gringa. These protected areas are administered by CONANP which, as a federal government agency, is a means by which the state maintains a direct role in biodiversity conservation. The cases of two of the recently declared protected areas in Oaxaca, the Reserva de la Biósfera Cuicatlán-Tehuacán and the Parque Nacional Huatulco are instructive.

The Cuicatlán-Tehuacán Biosphere Reserve, which crosses Oaxaca's northern border with the state of Puebla, was declared a protected area when the construction of a new major road connecting Oaxaca City with the industrial cities of

Puebla and Mexico City raised concerns that an area of outstanding biodiversity value was threatened. This concern came largely from biodiversity scientists based in Mexico City, and as such is a demonstration of their epistemic authority. However, many inhabitants of the area were neither aware of the declaration of this area as a Biosphere Reserve, nor of its biodiversity value. As Alejandro of *Campos Oaxaqueños* explained;

> The federal government declared it a reserve, but they never consulted the communities.

Huatulco National Park, on the Pacific Coast of Oaxaca, came into existence when the federal tourism agency, FONATUR, expropriated a stretch of coast to create the *Bahías de Huatulco* tourist complex. Of the expropriated land, 6000 ha was used to form part of Huatulco National Park. This was done to deflect environmental criticism of the tourist development; but it was not an initiative that was *a priori* supported by NGOs and, like the Cuicatlán-Tehuacán Biosphere Reserve, less still by the rural communities immediately affected.

The histories of these two areas, whilst not necessarily representative of all of Oaxaca's federally protected reserves and parks, demonstrate that the Mexican state remains willing to act independently of the non-governmental sector, and of the local people immediately affected by protected areas, in promoting biodiversity conservation. However, Oaxaca's NGOs do have a growing role as it is increasingly common for local conservation NGOs to be contracted to manage specific tasks within federal reserves. Such arrangements, which are more common in Chiapas where conservation NGOs are typically longer established, are increasingly common in Oaxaca where, for example, a local NGOs has been contracted to survey the flora of Huatulco National Park.

The histories of these two protected areas demonstrate a tendency for the biodiversity agenda to be enacted *ex cathedra* and regardless of the interpretative distance between the local people whose lives stand to be affected by that agenda and the concepts upon which it is based. At the same time, this autocratic approach to biodiversity conservation has failed to achieve the systematic conservation that conservation scientists have called for (Pressey *et al.* 1993). The Mexican state, it appears, has taken up the rhetoric of the biodiversity agenda but not necessarily the scientific prescriptions for its enactment.

7.4.3 The Non-Governmental Sector and Oaxacan Biodiversity Conservation

The World Wide Fund for Nature

The World Wide Fund for Nature opened its office in Oaxaca City in 1990, its first in Mexico, and since then has become one of the dominant actors in Oaxacan conservation. The choice of Oaxaca was made by WWF-US, and was influenced by a prioritisation exercise for Latin America and the Caribbean that pre-empted what is now known as WWF's Global 200 strategy (Olson & Dinerstein 1998). The Global 200 aims to represent the world's biodiversity by identifying outstanding ecoregions in all of the world's biomes and biogeographic realms. This was done by literature review, expert analysis and the use of GIS technologies. It is thus an attempt to guide systematic conservation planning at an international level using biological criteria. Remarkably, no less than four of the world's 200 priority regions are to be found partially or completely within Oaxaca: the 'Sierra Madre de Oaxaca pine-oak forests', 'Oaxacan montane forests', 'Balsas dry forests' within which Huatulco National Park is located, and the Tehuacán Valley matorral within which is found the Cuicatlán-Tehuacán Biosphere Reserve. The interest of this international NGO in Oaxaca can therefore be said to be a direct result of a broad scale, scientific biodiversity assessment exercise that was carried out remotely, far from the locality of interest. It is in this distanciated way the epistemic community of biodiversity scientists has extended the biodiversity agenda into Oaxaca.

WWF, which was founded in 1961, is a decentralised transnational organisation. National nodes in its network are semi-autonomous but WWF-International (based in Gland, Switzerland) co-ordinates strategy and controls some centralised resources for the network. This results in the various nodes, which include WWF-UK, WWF-US (based in Washington DC) having different approaches to common objectives. These differences have led to a significant divergence in approach by WWF-Oaxaca to that of the other WWF regional offices now established elsewhere in Mexico. Initially, funding for WWF's Oaxacan office came from the World Bank via WWF-US, however, the most important financial support was soon to come from the UK's Department for International Development (DFID) through its block grant to WWF-UK. WWF-UK's block grant from DFID is tied to a commitment to social development, the conservation of biodiversity alone is not enough, whereas WWF-US, which continues to be the main financial supporter of the other regional offices in Mexico, does not have this obligation. Thus part of the attraction of Oaxaca to

WWF-UK was that the social issues in this poorest of Mexican states were as pressing as the biodiversity issues and so their DFID block grant could be used to finance interventions in there. This shift in emphasis is significant because it portends an apparent impediment, that of a competing rural development agenda, to the seamless translation of the biodiversity agenda through the networks of international conservation.

The six conservation NGOs based in Oaxaca all received substantial funding in the early stages of their existence through WWF-Oaxaca. WWF-UK conceptualises its relationship with the local non-governmental sector as a form of partnership with civil society- NGOs are thus seen as a part of that wider 'civil society'. Whilst WWF does not have a single clear definition of what partnership should entail, the articulation of its objectives with partner organisations accords with three of the nine objectives which frame its relationship with DFID (Sarah Hutchinson, WWF-UK, pers comm).

- Civil society partner organisation able to deliver their own sustainable development agenda.
- Civil society engaging in international, national and regional processes of strategic frameworks of sustainable development.
- The mainstreaming of environment and poverty links.

Supporting NGOs in Oaxaca is therefore not just a means to promote biodiversity conservation, but an end in itself. Whilst the 'problem' the Global 200 seeks to address is expressed through what Bryant (2002) calls the 'biodiversity idiom' the solutions proposed have been interwoven with wider development objectives from an early stage of WWF's intervention in Oaxacan conservation.

It would be wrong to imply that WWF-UK's aim of strengthening Oaxaca's NGOs was an expedient born purely of DFID requirements. For one thing, WWF-UK has access to many sources of funding and has channelled funds from the European Union and the UK's National Lottery to WWF-Oaxaca. Furthermore, it is WWF-UK's policy always to work through local partner organisations regardless of the ultimate source of their funds. Hence, arriving in Oaxaca and finding a lack of suitable partner organisations WWF had to set about forming alliances with local NGOs, strengthening some and even creating others during the early to mid 1990s. As Isabela of WWF-Oaxaca put it:

> Part of the aims and objectives of the Oaxaca programme, the phase one, was precisely to strengthen local groups with whom we would be working

All six local conservation NGOs considered here have received substantial funding from WWF-UK, via the WWF-Oaxaca office, with the consequent extension of the network through which the biodiversity agenda could spread.

The shortage of suitable partners was not due to a lack of NGOs in Oaxaca. Mexico during the 1980s, like many Latin American countries, underwent a period of transition towards reduction in the size of the state and greater decentralisation, a period that has become known as *la apertura*, the opening up. One effect of this in Oaxaca, as elsewhere in Latin America (Aldaba *et al.* 2000), has been an increasing number of NGOs with a variety of aims and interests. Thus Oaxaca already had many NGOs but previously few directly addressed biodiversity conservation concerns. Another effect of this 'opening up' has been that many of the founders and senior staff of these NGOs, whose university educations in the past might have found them more secure employment in the state sector, have migrated to the non-governmental sector (Bebbington & Riddell 1997). Indeed, amongst all those interviewed, whether from conservation NGOs or from the government and academic sectors, non-Oaxacans were found to predominate with most having received tertiary education in the metropolitan areas of the country. Oaxacan conservation is not run by Oaxacans.

How partnership is enacted in Oaxacan biodiversity conservation is given further consideration in chapter 8. However, here it should be noted that the lack of a WWF-UK framework to guide interactions with partner organisations (which do not necessarily have to be NGOs) has the advantage of leaving considerable flexibility in partnership formation. In the case of Oaxaca, funding in the early 1990s was piecemeal and tied to certain projects and activities that WWF and the respective NGO had agreed were of mutual interest. Later internal reviews of that period carried out by WWF-UK suggest that success has been partial and that perhaps a more holistic approach to institutional strengthening with a focus on capacity building, rather than a series of collaborations, might have been more appropriate in order to leave each NGO in a stronger position as direct support from WWF has been reduced. Despite these reservations, WWF was undoubtedly instrumental in helping to form a vibrant and varied group of NGOs working for biodiversity conservation in Oaxaca.

FMCN

The Mexican institution of most importance in provision of financial support to Oaxaca's conservation NGOs has been the FMCN. It was constituted in 1994 as a government response to the 1992 Earth Summit in Rio de Janeiro and the subsequent CBD (Heywood & Iriondo 2003). It is also claimed that the founding of the FMCN was an attempt to placate environmental concerns raised by Mexico's entry into the North Atlantic Free Trade Agreement. With matching grants from the Mexican and the US Governments, a US$10 million fund was established and invested in the stock market. The proceeds, which are managed independently of government, are dispersed as project funding to civil society and research institutions throughout Mexico. FMCN's commitment to civil society goes beyond approving projects to aiding with project preparation. It recognises that many NGOs are not well versed in project preparation and it organises workshops for failed applicants to improve their proposals. In so doing it seeks to diversify the number of organisations through which it can support the conservation of Mexican biodiversity. Implicit in this is that FMCN seeks not simply to promote biodiversity conservation but, like WWF, to strengthen the role of non-governmental partners in biodiversity conservation.

The influence of science and scientists on conservation is evident in the FMCN. Its evaluation committee is formed by eminent Mexican conservation scientists and it counts biologists amongst its senior staff. Given its existence is a result of processes related to the CBD, and the continued influence of scientists on its mission, the FMCN can be said to be a product of the biodiversity agenda and a key articulation for its transmission to Mexico's non-governmental sector.

Local NGOs

Mexican organisations constituted as *asociaciones civiles* (civil associations) are recognised by the finance ministry as non-profit organisations not involved directly with party political activities and with autonomy from the state. Five of the six local NGOs discussed here are constituted as *asociaciones civiles*, the sixth, a *fidecomiso* is constituted as a non-profit trust fund managed independently of government. Hence , all six are considered to be NGOs both locally and in this investigation. All are relatively small, none having more than 10 full-time professional staff. The oldest traces its origins back to 1987, but did not become legally constituted as an *asociación civil* until 1991. Because of the small size and relative youth of the organisations, an interview with a senior staff member also meant an interview with a founder member in five of the six cases. A degree of territoriality has emerged in Oaxaca between these NGOs. This is in part due to interests in distinct ecosystems that date back to the NGOs' inceptions, yet even where they find themselves working in the same ecosystems there is a tendency for each to concentrate on different locations. This reduces competition for funding and ideas, but may also be the one of the causes of a lack of cooperation between NGOs.

Amongst the senior staff interviewed, all were university educated and most in the biological sciences. These scientific roots are reflected in the names (but not the pseudonyms) of three of the six NGOs, in which the words 'studies' (*estudios*) or 'investigations' (*investigaciones*) appear suggesting a research remit. In the biological sciences curricula of Mexico's main universities, like those of the northern universities, biodiversity has become the dominant conceptualisation of nature. In contrast, there is a wide appreciation amongst these NGOs, and other Oaxaca-based organisations involved in conservation, that biodiversity is not a term widely understood in rural areas (see chapter 4). A typical response when asked whether biodiversity is a concept widely referred to by rural people in Oaxaca was that of Mario of *Mesa Rural*:

> No, it's usually unrecognised, they [rural Oaxacans] talk of nature...They don't get to know the term as such, they are aware they have various resources, that they have water, trees, and from the forest they can get other services.

Thus Oaxaca's conservation NGOs represent the limits of the extension of understanding of biodiversity as a concept and yet they seek to extend the influence of the biodiversity agenda into rural Oaxaca where the concept is rarely understood.

By locating the principal offices in Oaxaca City, with its good communications to the capital and beyond, local NGOs are strategically placed- they are close to their international and national partner organisations but can still claim the rhetorical advantage of being closer to the important biodiversity found scattered across this mountainous state. They can 'speak' for Oaxaca's biodiversity in the national and international fora of the biodiversity agenda.

Within Oaxaca's non-governmental sector there is a general consensus that some benefits of conserving globally important biodiversity must accrue to local people, Héctor described the approach of *Naturaleza* thus:

> One of the goals we seek is not to conserve it [biodiversity] *per se*, but to conserve it as the patrimony of its owners.

The logic of this is that the local value of biodiversity must be realised. In general, the six local NGOs share, to a varying extent, a complex agenda that combines concerns for rural development with biodiversity conservation. This hybridity is achieved by proposing that biodiversity can be used as a resource. Roberto of *Ecodesarrollo* and Héctor, respectively, illustrate this:

> Rural development, as an aim of a large group of social organisations today, is about resources, resources like biodiversity.

> The interesting thing is how much you can convert this biodiversity into biological resources.

This position is informed by a pragmatism borne of the recognition that some of the most important areas of Oaxacan biodiversity cannot simply be emptied of people to create exclusionary protected areas. Thus social issues have forced their way on to the agenda of Oaxacan NGOs, compromising the seamless translation of the biodiversity agenda. A senior manager of FMCN described the process thus:

> What I see is that there are organisations that begin with a very biological vision and tend to modify their vision towards community development, because in Oaxaca it would seem to me that it would be very difficult not to fall into this transition, given that Oaxaca is so biologically and culturally diverse and that it is a state where there is extreme poverty.

Indicative of this is the experience of *Arbol Verde*, one of the NGOs most dedicated to biological research. They found that a floral survey of the region led inevitably to a consideration of the conservation status of fuelwood species used by local people and finally to the NGO supporting an initiative to create woodlots for that community. The important implication of this is that the NGOs of Oaxaca have to represent the interests of rural communities as well of the biodiversity of the state. Thus in each

NGO there can be detected one or more social agendas that dovetail with a common belief that conserving Oaxaca's biodiversity is *a good thing*. As was shown above, the commitment to social agendas is an important enabling mechanism for WWF-UK's commitment to Oaxaca.

NGOs do not necessarily work directly with members of rural communities but through community organisations or through municipal government structures, Hugo explained the attraction of this to *Ecobosque*:

> We have found that it is much easier to work with organisations, producer organisations; it's much easier than working with the open population in the community.

This inevitably opens a further interpretative distance between the epistemic community of the biodiversity agenda and the rural communities where it is intended that the agenda should be played out as it moves from international and national institutions to local NGOs and then to community or municipal organisations. The network becomes more complex, and the likelihood of translating the biodiversity actant unreconfigured into the practice of nature conservation becomes less probable. It is in this, and similar network spaces, that the epistemic authority of the biodiversity agenda competes with local development concerns, and it is a space that local NGOs occupy to their great advantage. Being in this space puts a considerable responsibility on local NGOs, one that international institutions promoting biodiversity conservation are happy to shift to them. This responsibility is that of translating the biodiversity agenda. Note the contrast with the state administered national parks discussed above which have in some cases circumvented the needs of rural Oaxacans in the state mediated enactment of this agenda.

7.5 Financial Diversification and Oaxacan NGOs

The most obvious means by which institutions representing the biodiversity agenda can exert influence on Oaxaca's local NGOs is though control of the financial resources on which those NGOs depend. The degree to which there exists a culture of dependency between major funders and NGOs is addressed here by a consideration of alternate sources of finance.

Although WWF and FMCN have been pre-eminent in funding Oaxaca's conservation NGOs since the mid-1990s, there has been a trend towards diversification of sources of finance. From its initial high in the first half of the 1990s, support to Oaxaca's conservation NGOs through WWF decreased towards the end of the millennium whilst the fall in stock values in the first years of the 21st century also drastically reduced the FMCN's disposable income. In response to this Oaxaca's NGOs have sought to diversify their funding partners with international donor organisations continuing to provide the majority of funding, a fact that attests to the strength of the 'imagined biogeography' of this biodiversity hotspot. Those donors include the Ford Foundation, the MacArthur Foundation, the British Council, *Fundación Panamericana*, Conservation International and the North American Wetlands Association. Mexican donor organisations have also been important with lesser but significant funding also being coming from government, through SEMARNAT and the Ministry of Social Development (SEDESOL), although these funding partners tend to contract the services of NGOs rather than fund projects proposed by those NGOs.

What this reduction in funding from WWF and FMCN has not stimulated is the pursuit of financial self-sustainability either through income generation (other than a small amount of contractual work) or through conversion to membership organisations.

Oaxaca's NGOs therefore remain dependent on funding partners, a phenomenon also noted amongst NGOs elsewhere (Townsend 1999). For Bebbington (1997) this raises serious questions as to whether NGOs' actions can be sustained. In the context of Oaxacan conservation it also raises doubts over delivery of the sustainable development objectives of the DFID/WWF-UK partnership. Diversification rather than sustainability has predominated and it cannot be argued that these NGOs are advancing towards the 'beyond aid scenario' of Aldaba (2000). They are liable to remain dependent on ideas, concepts and approaches imported from outside of Oaxaca and to continue to have to compete with one and other for the limited funds available.

The reduced dependence of Oaxaca's NGOs on WWF is not necessarily considered a bad thing by those NGOs, as the experience of *Arbol Verde*, recounted by Yolanda, shows:

So we stopped working with WWF...because of a whole series of problems that were happening, like an old couple we had started to have problems, problems, problems...we decided to divorce, divorce was necessary.

The diversification of funding sources has at least created some space in which local NGOs can manoeuvre by giving them a choice of donor agendas to respond to. That donors, such as The Ford Foundation, whose interests lie as much in rural development as they do in biodiversity conservation are now prominent funders of Oaxaca's conservation NGOs may be interpreted as both a cause and an effect of rural development becoming increasingly important in the activities of several, if not all, of these NGOs. Thus as the biodiversity conservation network has extended, it has increasingly become enmeshed with rural development networks.

However, diversification in funding does not preclude the possibility that some of Oaxaca's NGOs might ultimately disappear. In the neighbouring state of Chiapas, where NGO involvement in conservation has a longer history, biodiversity conservation has become consolidated around fewer, larger NGOs than once worked there, and currently work in Oaxaca.

7.6 The Biological Sciences in Oaxaca

Given that science and scientific practices are central to biodiversity conservation, the practice of science in Oaxacan institutions offers a potential means of extending the biodiversity agenda into Oaxaca independently of local and international NGOs. Institutions of relevance to the advancement of conservation sciences in Oaxaca include the semi-autonomous State Ecology Institute (IEE) and the Oaxacan Ethnobotanic Garden. Biologists, typically educated outside of Oaxaca, are well represented in these institutions. However, they currently have only indirect influence on *in situ* conservation and, like other Oaxacan institutions involved in research, are considered inferior to the research institutions, particularly the universities, of Mexico's metropolitan centres. Oaxaca's status as one of the lowest income states of Mexico is reflected in the small contribution Oaxaca has made to Mexico's scientific advancement. The resources that establish scientific facts about biodiversity in Oaxaca remain largely under the control of the metropolitan centres where Mexico's biodiversity scientists are concentrated and, inevitably, botanical and zoological collections made in Oaxaca have accumulated in the herbaria and museums of Mexico City. The work done by these institutions, whilst being *about*

Oaxaca, is *for* national and international audiences and not responsive to the practice of nature conservation in Oaxaca. Over the past 25 years there has be some change in this. In 1981 the Agricultural Technology Institute of Oaxaca (ITAO) was established to teach agriculture-related disciplines to degree level, and in 1983 and the tertiary education college the Centre for Interdisciplinary Research for Integrated Rural Development (CIIDIR) was opened and manages the recently established Oaxacan State Herbarium. Both institutions teach predominantly Oaxacan students but their graduates have yet to establish a significant presence in the conservation community.

In the opinion of Isabela of the WWF office in Oaxaca City, the apparent lack of local capacity in conservation sciences has not entirely been compensated for by the influx of staff with biological degrees. WWF-Oaxaca has felt the need to call on external expertise to strengthen capacity and validate results. This lack of capacity, it is felt, has hindered its ability to engage with international actors in biodiversity conservation:

> Being as how we are not a local organisation, we respond to an international network, we cannot leave things so weak; we had to provide support so that things really have validity and credibility when we comment on whatever situation.

Expert guidance of high international scientific status on conservation issues is therefore highly valued by transnational conservation agencies and increasingly these international actors employ their own biodiversity scientists (da Fonseca 2003). The WWF-Mexico office exemplifies this dependence on scientific expertise:

> But it [scientific guidance] ranges from the most basic which is calling together the experts that exist on that region...it might be national experts, it might be international experts ...and getting people to draw lines on maps saying that this is an area that is very important. From there it can extend up to various levels...of rigour in terms of carrying out scientific investigation, compiling the databases that have all this information, looking at GIS to see how these areas overlap.

Thus there are important actors who do not doubt the contribution scientific understandings might make to the practice of Oaxacan nature conservation, however, that contribution has yet to materialise in the form of the 'systematic' or 'scientific' selection and management of reserves. Whether this will change as Oaxaca's research capacity continues to expand remains to be seen but it will be at least partially dependent on Oaxacan graduates remaining in, or returning to, the state.

7.7 Discussion: Networks and Translations

It has been shown that outside of CONANP administered protected areas, the Mexican federal government has limited its role in nature conservation to the provision of funds (FMCN), information (CONABIO) and regulation (SEMARNAT). This has had the effect of creating a space for international and local NGOs to become dominant actors in the translation of the biodiversity agenda. Many of the local NGOs have, however, institutional objectives that go beyond nature conservation and they have sought diverse funding sources to match. Scientific approaches and practices to nature conservation are therefore not necessarily considered essential by these NGOs.

The commitment of transnational NGOs, and to a lesser extent governmental agencies, to Oaxaca's local NGOs is justified by the assumed ability of local organisations to better understand Oaxaca's threatened habitats and their human occupants than can their more distant funding partners. This is an example of what Mohan and Stokke (2000) describe as the emergence of the local as a site of notional empowerment. Thus, instead of following the prescriptions of modern biodiversity science, they direct a significant proportion of the resources available to them to what they perceive as the more urgent needs of rural development. This may of course have many indirect benefits for biodiversity but reduces the importance of scientifically mediated systematic conservation.

Biodiversity conservation finds itself in a paradoxical position. Never has so much international coordination and funding been brought to bear on the problem of loss of diversity than it has today and never has so much public sympathy and understanding of environmental issues been apparent. Yet pessimism pervades the stories biodiversity scientists continue to tell about the loss of biodiversity (Cincotta *et al.* 2000; Freyfogle 2003; Johns 2003). Perhaps some of this can be attributed to the desire of biodiversity scientists to maintain public and political interest in their cause, and hence maintain funding levels for their activities. Yet even the most sceptical acknowledge that the loss of biodiversity continues to be a real concern (e.g. Lomborg 2001). Hence the paradox, all this good will and apparently there is so little to show for it.

That the epistemic and financial authority of the biodiversity agenda has not led to well planned and managed reserve networks in Oaxaca supports Law's (1994 p. 15) contention that we cannot view the world as:

> A lot of social products moving around in structural pipes and containers that were put in place beforehand.

Instead it is the result of orderings and systems that 'jostle' together to 'generate the social'. Conservation in Oaxaca is a network of diverse actors and actants that includes science and scientists; local, national and transnational institutions; rural development and biodiversity conservation agendas. Each promotes or represents different interests. In Murdoch's (1998) typology of networks of prescription and networks of negotiation, biodiversity conservation is an actant in a network of negotiation. The network stretches over too great a cultural and geographic distance and encompasses too many competing interests for the translation of the biodiversity agenda to be achieved perfectly.

Latour (2004 p. 235) sees the sciences as a set of practices for 'socialising non-humans' and in a short space of time scientists have been very successful in socializing the non-human actant known as biodiversity, it now being the subject of international agreements, university curricula and wider environmental discourses. As the actant has travelled to the hotspots of Oaxaca it has, perhaps inevitably, undergone a different process of socialisation, it has been translated. Translation consists of 'combining two hitherto different interests...to form a single composite goal' (Latour 1999 p. 88) and therefore biodiversity has, on arrival in rural Oaxaca, kept some of the characteristics of the globally important public good of the biodiversity agenda but taken on the additional characteristics of a local resource needed economic development. WWF-UK has attempted to influence Oaxacan conservation principally through institution building and in this it has had success in supporting Oaxaca's conservation NGOs. These local NGOs have maintained an association with the biodiversity agenda not least because of the funding opportunities it allows. However, only selectively have they allowed it to translate to 'the field' where nature is not represented as biodiversity and rural development concerns have to be accommodated. This translation is further discussed in chapter 8.

The ability of Oaxaca's NGOs to control the agenda has much to do with the rhetorical advantage conferred on them not only by the assumed advantage of 'the local' that is current in development and conservation discourses (Mohan & Stokke 2000) but also by the 'need' of institutions such as WWF for partnership. That local NGOs have been able to use both their location and scientific discourse to their strategic advantage is at odds with critiques of conservation elsewhere in Mexico. Both Haenn (1999), in reference to the Calukmul Biosphere Reserve in the Yucatán, and Harvey (2001) in reference to the Lacandon rainforest of Chiapas, conceptualise biodiversity conservation as an imposition of the powerful, that is national and international interests, on the less powerful, that is the local. This is achieved through what Norton (1997 p. 245) describes as the '*ex cathedra* pronouncements of the environmental expert'. Oaxaca's NGO community demonstrates that the biodiversity agenda can be contested and reconfigured locally, and perhaps always is. The network of negotiation formed is characterised by a degree of local autonomy and perhaps represents a step towards Brown's (2003 p. 91) goal of creating:

> New institutions for conservation and development that are flexible and adaptable and which are able to manage complex ecological systems and accommodate diverse stakeholder interests and values.

By conceptualising Oaxacan conservation as a network of negotiation some relief may also be had from the pessimism that seems common amongst conservation scientists. In one of the priority ecosystems of the state, the seasonal dry forests of the coastal lowlands, real progress can be boasted both with respect to knowledge of the biodiversity of that region (Salas-Morales *et al.* 2003) and, as will be argued in chapter 9, with respect to the area of forest under conservation-sympathetic management. That scientific prescriptions have not been followed should not be lamented but expected and accepted as a price worth paying for addressing the loss of global value through local action.

The international conservation community has created, from a distance, interest in Oaxaca and has acted to extend its agenda into Oaxaca. However, in the very act of extending its authority, authority has been lost, depending as it does on a group of locally based NGOs for the implementation of biodiversity conservation. Eden (1998 p. 426) argues that a linear model 'that assumes a one way flow of information – from science to policy to society' must be replaced by one of 'mutual negotiation and (re)construction of environmental knowledge between scientists and policy-makers'.

This seems to be what has happened in Oaxaca, except that negotiation between policy makers and practitioners has been shown to be crucial to the process.

However, there remains an important caveat to this potentially optimistic analysis of power relations in Oaxacan conservation. Oaxaca's NGOs are not grassroots organisations but are themselves populated by outsiders who have brought with them to Oaxaca their own agendas that are distant both in language and in origin from the understandings of Oaxaca's rural populations. The degree to which these agendas coincide with, or are informed by, those of the inhabitants of Oaxaca's biodiversity hotspots thus becomes the crucial question for Oaxacan conservation. Massey (1991) has pointed out that relative mobility and power over communication, of the sort that Oaxaca's conservation NGOs have over their client groups, can lead the latter into 'spatial imprisonment'. The close working relationships these NGOs have developed with the community-based organisations may prevent this from happening. However, it cannot be assumed by national and transnational conservation agencies that partnership with 'local' organisations is the same as partnership with those people most closely associated with the Oaxaca's extraordinary biodiversity.

Chapter 8: Partnership and Landscape in Oaxacan Conservation

8.1 Introduction

In its relatively short life, *biodiversity* has come to be interpreted in different ways and in different contexts (Harper & Hawksworth 1994; Wilson 1997). For Sarkar (2002 p. 137) it can be intuitively construed to mean 'all of biology', whilst for Gaston (1996a) it is, more narrowly, 'the biology of numbers and difference'. Fairhead and Leach (2002 p. 102), draw attention to its discursive role by describing it as a 'central organising concept in international environmental debate' whilst Callicot *et al.* (1999 p. 22) criticise the lack of clarity surrounding its meaning, describing it as one of 'a plethora of ill defined normative concepts in conservation biology'. Perhaps, however, Takacs (1996 p. 99) makes the greatest claim for biodiversity's central and fundamental role in modern conservation biology when he describes it as a term that

> [M]akes concrete - and promotes action on behalf of - a way of being, a way of thinking, a way of feeling, and a way of perceiving the world.

Despite these multiple definitions and contexts a consensus has emerged that the conservation of biodiversity is one of the major environmental challenges facing humanity. This consensus has in turn found expression in international politics through global policy and financial instruments such as the Convention on Biological Diversity and the Global Environment Facility (Dalton 2000; Raustiala 1997b; Royal Society 2003). Thus the biodiversity agenda (see chapter 2) can now be considered part of the 'macro-social' (Law 1992); a stabilised interaction, or discourse, played out between scientists and policy makers who operate both at international and national scales.

Institutions, including many in the non-governmental sector, have adapted to this discourse to form a transnational network that, despite its diversity:

> [S]ynchronises behaviour, outlook and language along common lines all around the world (Mawdsley *et al.* 2002 p. 4).

In contrast to this internationalisation of biodiversity conservation politics, the practice of biodiversity conservation remains localised and site specific. Particular places, known variously as hotspots (Mittermeier *et al.* 1998) or centres of diversity (Heywood 1995), that are often found in mid- and lower income countries have become the foci of influence for this transnational network. In order to put into practice this new, global politics the idea of partnerships has been invoked by actors in this network to encompass the variety of relationships that exists between northern and southern institutions. Mawdsley *et al.* (2002) go as far as to argue that, in the wider context of development in lower income countries, the discourse of partnership is critical to the whole realisation of an emergent transnational community. Typically, this relationship is conceived of as one in which the former provides the funds, and often sets the agenda, whilst the latter provides the local knowledge considered necessary for successful implementation.

Lister (2000 p. 228) summarising the promise of partnership, in the wider context of development, suggests:

> North-South partnerships are currently seen to enable more efficient use of scarce resources, increased sustainability and improved beneficiary participation in development activities. Furthermore, it is thought that the creation of synergy through partnership produces results that partners could not obtain without collaboration.

Given such a positive theorisation of the concept, it is not surprising that partnership has been readily incorporated into the rhetoric and practice of non-governmental organisations (NGOs), including those promoting biodiversity conservation. Partnership, between international conservation organisations and local non governmental actors, is now an established trend in biodiversity conservation (e.g. Hulme & Murphree 1999; Murombedzi 1999). Indeed transnational biodiversity conservation institutions can be said to be heavily dependent on partnership to put their biodiversity agenda into practice.

Partnership between institutions as an organizing principle for north-south relations is highly contested. It recognises the local as a site of empowerment and yet often does so without being critical of the underlying assumptions and dangers of what Mohan and Stokke (2000) describes as 'localism'. In contrast, it has been argued that invoking partnership as an organising concept is a means of placing a veneer of equality on what are in reality unequal relationships between 'dominant' transnational institutions and their 'submissive' southern counterparts. In this latter interpretation, concepts are transferred from north to south (Tvedt 1998) in a dependency inducing relationship that is postcolonial in character (Townsend *et al.* 2002). Here consideration is given to the power relations of partnership in the particular circumstances of biodiversity conservation as practiced by international and national organisations and their local NGO partners in Oaxaca, Mexico.

The point of entry for evaluating these partnerships is the concept of biodiversity. It might be assumed that a partnership would require a shared understanding of the concepts around which it is organised and that, hence, successful biodiversity conservation would require substantial consensus between partners on the meaning of biodiversity. Indeed Callicott argues that in the absence of a clear understanding of, and consensus regarding the 'normative idea' of biodiversity, 'conservation efforts will be confused and confounded' (1999 p. 23). How local NGOs in southern Mexico interpret the 'northern' concept of biodiversity, how it is (re)configured in their rhetoric and their work are here discussed in the context relations with their donor partners. The institutional environment in which Oaxacan conservation is enacted is briefly described. The preferred scientific/northern configuration of biodiversity as diversity of *species* is then contrasted with an emerging reconfiguration of biodiversity as diversity of *land use practices*, that is, biodiversity as a landscape phenomenon. This is contextualised by brief discussion of the socio-political applications of the landscape idea in Mexican history and then by consideration of the relationship between the use of landscape by Oaxaca's NGOs and the scientific disciplines of landscape ecology and agro-ecology. The use of geographic information systems is shown to be important in this reconfiguration and it is argued that this translation of the biodiversity idea is political in that it is an attempt to rework global politics for a local setting. The significance of this reconfiguration is then discussed in relation to the meaning of partnership between local NGOs and the organisations that fund them.

These issues are explored through an interview survey carried out in Oaxaca and Mexico City in 2002 and 2003. It consisted of 18 semi-structured interviews with members of a variety of institutions active in, or influential on, Oaxacan conservation. Included were governmental (state and federal), academic, and non-governmental institutions. Whilst all interviews proved to be relevant, most central to the issues raised here were interviews with members of a group of six NGOs based in Oaxaca and one of their principal funding partners, The World Wide Fund, for Nature (WWF) which maintains a main office in Mexico City and a regional office in Oaxaca City.

8.2 The Institutional Context of Oaxacan Conservation

Over the last decade, the conservation of Oaxaca's biological resources has become increasingly mediated by the non-governmental sector. Funding has been supplied from and via international non-governmental donors such as The Ford Foundation, the MacArthur Foundation, Conservation International and the World Wide Fund for Nature (WWF) and national bodies such as the Mexican Fund for Nature Conservation (FMCN), to the group of conservation-orientated NGOs based in the State capital, Oaxaca City. The institutional aims of these local NGOs combine, with varying degrees of emphasis, the conservation of Oaxaca's natural diversity with the social and economic development of Oaxaca's rural populations. These aims are implemented through projects that reach out to the peripheries of the state in the mountains and coastal areas where that rural population, one of the poorest and most marginalized in Mexico, lives.

Oaxaca's conservation NGOs are likely to remain financially subordinate, even in the diversified relationships described in chapter 7. Their aims and objectives inevitably have to accommodate those of their funding partners. The question therefore arises as to how that accommodation is made in light of the epistemic authority of biodiversity agenda.

The growth of partnership agreements between international and local NGOs, coupled with state contraction which has in recent years seen Mexico's ministry for environment and natural resources (SEMARNAT) withdraw from an implementation role to a largely regulatory one, has meant that locally based NGOs are now amongst the most important of actors in Oaxacan conservation. Despite the broad similarity in objectives among Oaxacan NGOs there is a geographical separation in their activities with little overlap between the areas and communities in which each seeks to apply its influence. These organisations are all relatively small, none having a staff of more than ten professionals. Most of their senior staff are university educated, and a have arrived in Oaxaca from the metropolitan centres of Mexico where they were born, raised and educated. Hence their influences and agendas are not be assumed to be those of the rural Oaxacans whose environment they seek to influence.

Organisations and interviewees quoted directly are given here although pseudonyms are used for all individuals and for the six local NGOs.

- *Naturaleza* promotes integrated development and conservation activities, principally in the south of Oaxaca. Hector, a biologist, is its director.
- *Mesa-Rural* began as a 'round table' organisation providing a space for various actors in Oaxacan conservation to share ideas and experiences. Now, it is independent of those other organisations and works largely with geographic information systems (GIS) on land use planning projects for rural communities. Mario, the only Oaxacan interviewed here, is its director
- *Arbol Verde* is one of the longest established NGOs in Oaxaca, dating back to 1987. Its director is Yolanda, a biologist. They combine research with rural development and have a strong GIS capability.
- *Ecodesarrollo* is led by Roberto and has various interests in Oaxaca and beyond, including sustainable forestry and agricultural research, biodiversity conservation and rural development. Roberto, its director, is a biologist.
- *Ecología* works on various rural development initiatives with a concern for biodiversity conservation particularly through watershed management projects. Its director, Carlos, is a biologist.
- *Ecobosque* is led by Hugo, an anthropologist, and oversees projects on forest conservation and management in the mountainous areas of Oaxaca.
- WWF has had an office in Oaxaca for nearly a decade, and one of its senior staff is Isabela is a long-standing manager of projects and relations with Oaxaca's NGOs. Sally was, at the time, of interview an interim manager in the Mexico City office, on secondment from WWF-UK.

The attraction of Oaxaca as a place to 'do' nature conservation is that it has been identified as being one of the most biodiverse areas (Ceballos *et al.* 1998; Salas-Morales *et al.* 2003; Velázquez *et al.* 2003) of this biodiverse country (Mittermeier 1988; Ramamoorthy *et al.* 1993). Modern nature conservation is now inextricably linked with the concept of biodiversity and the staff of Oaxaca's conservation NGOs have become, through their university educations, through the popular media and through interactions with their funding partners, conversant with this concept. This is despite the term's origins being in the research institutions of North America (Harper

& Hawksworth 1994; Wilson 1997), which are both culturally and geographically distant from rural Oaxaca. Hence the significance of the central theme discussed here, how the concept of biodiversity is translated from scientific academia into practice.

8.3 Biodiversity and Biodiversity Assessment in Oaxaca

Despite the diverse contexts in which biodiversity is conceptualised, a consensus has emerged, particularly amongst biodiversity scientists working for northern institutions, around its partitioning into three levels, the 'holy trinity' of genetic (within species), species and ecosystem diversity (Sarkar 2002 p. 139). This conceptualisation is now enshrined in the Convention on Biological Diversity (CBD) (CBD 2001):

> "Biological diversity" means the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems.

This definition is essentially of the 'compositionalist glossary' of Calicott *et al.* (Callicott *et al.* 1999) with its concern for what makes up biodiversity rather than the functionalist glossary which concerns itself with what biodiversity does. The WWF shares an essentially similar, if more populist, definition:

> Biodiversity is the incredible variety of life on Earth - everything from the tiniest microbes to the tallest trees, from creatures that spend their entire lives deep in the ocean to those that are anchored in the soil of the Earth's crust. Genes, species, and habitats -- and even the fantastic range and expression of human culture -- are all part of our planet's biodiversity. (http://worldwildlife.org/windows/overview.cfm)

Thus, WWF can be seen as a 'transmission channel' (Townsend *et al.* 2002 p. 830) by which this compositionalist construction of nature is carried to their local NGO partners in Oaxaca.

Biodiversity assessments, that is exercises in locating and prioritising biodiversity, relate to the compostionalist glossary in their use of species and ecosystems as their ontological foundations, (genes are too difficult to count under most circumstances). For biodiversity scientists the ideal is that biodiversity assessment should set priorities in conservation, determining what should be the subject of conservation. Estimates of numbers of species, or numbers of rare species, or measures of the uniqueness of ecosystems within a given area are typically combined with some estimate of threat to assess and define priority areas for conservation (e.g. Dinerstein *et al.* 1995; Myers *et al.* 2000; Pressey *et al.* 2003), and regional assessments of this sort have stimulated interest in conservation in Oaxaca. In contrast, within Oaxaca the rigorous application of biodiversity assessment has not been used to guide local decision-making by local NGOs. Given that consensus around the definition of biodiversity might be expected to be a prerequisite of the use of biodiversity assessment, what is considered here is whether Oaxaca's local NGOs chose a different ontological foundation to that of the CBD and WWF in constructing nature with the possible consequence that biodiversity assessment would be rendered obsolete.

8.4 Biodiversity and Rural Development

Biodiversity conservation is not the unique concern of the NGOs of Oaxaca. Despite their close relation with national and international partners that promote biodiversity conservation, such as WWF and the Mexican Fund for the Conservation of Nature-(FMCN), their work is framed by a concern for rural development. Carlos of *Ecología* elucidated:

> This notion of conservation as simultaneously society and nature is a basic element, we believe that they cannot be separated, especially in Oaxaca they cannot be separated, you cannot talk of conservation without people, you cannot talk about social wellbeing without nature.

The 'social wellbeing' of Oaxaca's rural communities depends on more than just the conservation of the maximum number of species possible in a given area. To bring together the separate discourses of rural development and biodiversity conservation, which in Oaxaca usually equates to forest conservation, the practices of agriculture and forestry have to be reconstructed rhetorically to become biodiversity conservation practices. As Hugo of *Ecobosque* explained:

> Well, biodiversity fits, I think, just as much within the efforts being made to conserve spaces for forestry and for agriculture- given the quantity of life that you find in them.

This conflation of conservation and rural development results in a reworking of the boundaries that circumscribe biodiversity conservation in order to bring entire landscapes under its influenced, rather than the remaining forests alone, and this is encapsulated by the following example in which landscape (*paisaje*) is the preferred spatial conceptualisation with which to do this.

At the eastern limits of Oaxaca, in the mountains that cross the state border with Chiapas, is a region known as the Chimalapas. The region is still densely forested with one of Mexico's few areas of tropical rainforest. The Chimalapas have attracted particular interest because of both high total species diversity, as is typical of rainforests, and high concentrations of rare species. The epistemic community of biodiversity scientists have defined the region as an area of outstandingly important biodiversity, and one that should receive the attention of the wider conservation community (Ceballos *et al.* 1998; Wendt 1993). They have had success, the region is now one of WWF-Oaxaca's priority ecosystems and part of it is now protected in the federally managed La Gringa National Park. However, the rationale of importance defined by numbers of species and numbers of endemics is not necessarily shared by all of Oaxaca's local NGOs. The following was recounted by Roberto of *Ecodesarrollo* in which he compares the Chimalapas with Tuxtepec, an area in the northeast of the state towards the border with Veracruz (emphasis is added):

> The method has basically been to look at satellite images and see where there is no fragmentation, which is misleading- biodiversity gets overlooked. If we compare the conditions in the Chimalapas (which has variation in altitude, aspect and the rest, but in general is a more or less continuous block of vegetation) with that of Tuxtepec where there is much fragmentation, well its obvious that the landscape diversity is greater in Tuxtepec than in the Chimalapas....in Tuxtepec there are pastures, fallows, there are cultivated areas, small patches of forest or other types of woodland, there is a *landscape diversity*

Here the well-preserved tropical forest is reinterpreted as just one element of a diverse agricultural landscape. This entails a reconfiguration of biodiversity that is significantly different to that of the conservation biologist. Species diversity is not a relevant factor here, and no judgement is offered on the number or rarity of species in an area (cf Ceballos *et al.* 1998; Wendt 1993). It does not correspond to ecosystem or ecoregion diversity (Dinerstein *et al.* 1995) as no consideration is given to the uniqueness or threat status of the Tuxtepec ecosystem compared to that of the Chimalapas or other ecosystems. Instead it is variation between land use practices that is the manifestation of biological diversity and, notably, no distinction is made between anthropogenically altered land uses such as pastures and more 'natural' land uses such as forests. The result is a reconfiguration of biodiversity as both component and product of rural land use practices. Hector of *Naturaleza* put it thus:

> When you talk about managing biodiversity its about more than managing species, it is about managing landscapes.

This reconfiguration of biodiversity, here referred to as *biodiversity as landscape*, brings agricultural land use practices to the fore in Oaxacan conservation politics and must inevitably compete with the view of biodiversity as the variation between species in which biodiversity assessment and systematic conservation planning are grounded (Pressey *et al.* 1993).

It is axiomatic in northern conservation thinking that the most 'natural' and 'undisturbed' habitat types should be prioritised for conservation as undisturbed areas are becoming increasingly rare. It can be assumed that in much of Oaxaca, forests represent the most natural component of the landscape, the one that would have existed prior to anthropogenic alteration. In proposing that other land uses might deserve equal priority, Oaxaca's NGOs contest this focus on the unaltered forest and propose that other components of the landscape that are in themselves in need of conservation. Carlos of *Ecología* gave this example:

> The traditional *milpa* [maize field] is an example of the promotion and active protection of biodiversity, despite what is usually thought... Another aspect that has been studied is the direct creation of cultivated species, the fact that we have 100 races of maize in Mexico results from biodiversity produced by cultivation.

However, this reconfiguration is about more than just a concern for biodiversity, it is essential to NGOs in the accommodation of biodiversity conservation objectives with their rural development agendas. The prescriptions of the biodiversity scientist might require the restriction of activities that reduce forest cover and quality (a quality that is here gauged by the narrow metric of species diversity) whilst the manipulation and clearing of forest, albeit temporarily for rotational cropping, are established land use practices in agrarian Oaxaca. By making land use practices other than forest equally important the potential conflict between agricultural development and biodiversity conservation disappears.

In admitting these anthropogenic components, humans are relocated to a more central position in conservation politics and this accommodation is given further justification by the contention that to alter a diverse landscape in favour of more uniform one would represent a high risk strategy, as Roberto of *Ecodesarrollo* explained:

> [W]e feel that if we recommend that people should move towards low diversity systems, we are exposing them to too much risk, so to maintain a certain level of diversity at whatever level is first and foremost a means of reducing exposure [to risk].

Writing in one of the 'in-house' journals of conservation science *Conservation Biology*, Callicott *et al.* (1999 p. 24) contrast the compositionalist and the functionalist schools of conservation biology. They contend that compositionalists perceive the world through entities such as species and populations, i.e. the northern configuration of biodiversity, and in so doing separate humans from nature. Functionalists have a more process-orientated view that encompasses energy flows and ecosystem ecology into which humans are integrated. The emerging interpretation of biodiversity as a landscape phenomenon in Oaxaca suggest a translation from the dominant biodiversity/compositionalist view to a more landscape/functionalist approach. The tension between these 'schools' is one that has surfaced elsewhere, particularly where anthropogenically altered landscapes are the focus of conservation. In the context of conservation in the UK, Adams (1996 p. 135) describes a 'great divide' between landscape and nature conservation.

This is more than just a rhetorical reconfiguration as it finds expression in the practices of Oaxaca's NGOs. One community-based programme operating in the south of the state and overseen by *Naturelaza* is entitled the 'Community System for the Protection and Management of Biodiversity' (SICOBI). This programme is essentially about the protection of a watershed and is composed of four basic sub-programmes, only one of which 'biodiversity protection' justifies the title. The other sub-programmes concern the sustainability of various land use practices and include, 'Community agroforestry', 'Sustainable coffee production' and 'forest restoration and silviculture'. These might incidentally lead to the protection of biodiversity but they are not, in terms that biodiversity scientist would understand, biodiversity conservation programmes. Nonetheless, it was decided that biodiversity should be part of the title of the project, justified by the variety of land uses it seeks to influence, and The Ford Foundation were prepared to fund it. The reconfiguration of biodiversity as landscape allows the NGOs to maintain the rhetorical advantage of biodiversity conservation whilst practicing rural development.

8.5 Modern Conservation and the Landscape Tradition.

Landscape is associated with a long tradition of cultural analysis and the translation of biodiversity that is emerging in Oaxaca needs to be put into this cultural and historical context. Ingold (1986 p. 153) maintains that landscape is no less than the concept that distinguishes intelligent beings from the unintelligent. Its use demonstrates that environment is perceived as a set of essences waiting to be organised into a project. In particular, he argues that it is the cultivator, as opposed to the hunter-gatherer, who 'appropriates the land in plots...within a landscape'. Here the idea of landscape brings humans and nature together, as it does for Oaxaca's NGOs.

Cosgrove (1985) suggests that European notion of landscape as the embodiment of a kind of romantic holism was a reaction to the scientific revolution and its Cartesian division of subject and object that was to result in an increasingly mechanistic way of interpreting nature. Here there is a parallel with the move from reductionist compositionalism to holistic functionalism. The idea of landscape that has emerged in the North has associations with, variously, the taming of nature, romanticism and the management of the rural idyll (Coates 1998). That idyll, of course, has little to do with the realities of life in modern rural Oaxaca. Landscape as an artistic concept was also, according to Cosgrove (1985 p. 46): 'Over much of its history, closely bound up with the practical appropriation of space' and in this respect was associated with colonial expansion. Landscape had a major impact on Mexico's colonial history. Sluyter (1999) proposes, with particular reference to the Mexican Gulf port of Veracruz, that Mexico's colonial elites reconfigured the landscape through oppositional categorisation: cultivated vs. wilderness; civilised vs. savage; and social vs. natural. These binaries served the colonisers well as they brought the landscape under European methods of husbandry that, particularly in the case of livestock farming, were completely alien to indigenous inhabitants. Each binary was used to demonstrate progress with the desirable 'cultivate', 'civilised' and 'social' opposing the undesirable 'wilderness', 'savage' and 'natural'. Thus conceptual transformations were closely linked to material transformations as livestock farming grew and European cropping systems spread. However, prior to Spanish colonization what was to become Mexico was already composed of transformed and diverse landscapes thus these binaries did not serve to accurately describe these material transformations so much as conceptualise and legitimise the process of colonisation.

Oaxaca's landscapes have been continuously reconstructed over time, materially and conceptually. Livestock has been introduced, indigo production boomed in the colonial era and then declined, coffee was then introduced to the higher altitudes and indigenous tenure systems (*ejidos*) have been alternately undermined by colonisation and reasserted following the Mexican revolution. Echoes of each preceding management regime have survived into the next and one result of which is that it is no longer easy to determine, even in the most under-populated and forested areas, what pristine vegetation would be in most of Oaxaca. There is no single historical point of reference to be taken from this 'palimpsest' (Bender 1993; Crang 1998) neither for northern biodiversity scientists with their aim of maximising biodiversity, nor for Oaxaca's NGOs seeking to integrate nature conservation and rural development. Pristine nature is gone (Denevan 1992; Gómez-Pompa & Kaus 1992; Sprugel 1991) and its value of what remains is dependent upon the constructions with which it is interpreted. There is therefore little new about actors from outside of rural Oaxaca, which now include local NGO's based in Oaxaca City, making competing claims for the reconfiguration of Oaxaca's rural landscapes. What is different here is that Oaxaca's NGOs are attempting a reconfiguration to incorporate another culturally contestable but much newer concept, that of biodiversity.

Oaxaca's NGOs could therefore stand accused of being the promoters of another postcolonial reconfiguration of Oaxaca's landscape, and as Young points out (2003 p. 140): 'No act of translation takes place in an entirely neutral space of absolute equality'. However, what might be added to that statement is that choosing to make no translation/reconfiguration is an equally value laden enterprise. This is especially so given the context of Oaxaca's rural population which is, and ever was, experiencing simultaneous reconfigurations of the rural economy as families, once reliant on farming, increasingly work in the tourist industry and abroad. Oaxaca's NGOs cannot prevent the re-territorialisation of Oaxaca as a biodiversity hotspot, or several biodiversity hotspots. For contemporary Oaxaca enquiry is better directed away from asking how a landscape came to be and towards examination of what it is to be used for (Rose 2002). Whilst NGOs are implicated in the transmission of the biodiversity agenda through their relations with their funding partners, they are paradoxically a site of its subversion. By the creation of a place for agricultural landscapes in the discourse of biodiversity conservation, Oaxaca's rural population may stand to gain rather than lose through Oaxaca's new status. Protected areas, the most obvious and at times the most culturally inappropriate manifestation of that agenda, may come to be seen as only one of many conservation options in Oaxaca.

8.6 Scientific Justification of Biodiversity as Landscape

Landscape ecology and Agro-ecology

To contextualise the use of landscape in Oaxacan conservation it is necessary to consider its relation to two sub-disciplines within applied biology that it appears to resemble; landscape ecology and agro-ecology.

The science of landscape ecology is relatively new, having emerged in central Europe post World War Two (Farina 1998). It recognises that ecosystems are composed of patches of different habitat types, and it explores the biotic interactions between patches in mosaic landscapes. The relationship between this science and the reconfiguration of biodiversity as a function of landscape is important because landscape ecology has the potential to lend scientific authority to the view of biodiversity being proffered by some of Oaxaca's NGOs. Beyond the natural patchiness of ecosystems, landscape ecology has become prominent in the conservation sciences as human activities have further fragmented landscapes (Hunter 1996) to the point at which conservation is now very much about the management and mitigation of the effects of anthropogenic fragmentation on the survival of species.

The development of Island Biogeographic Theory (MacArthur & Wilson 1967), gave landscape ecology a theoretical foundation for predicting the numbers of species that islands could sustain based largely on the simple metrics of island size and distance from continental sources of immigrant populations. The relevance of this to conservation biology was immediately recognised, with nature reserves being conceptually reconfigured as 'islands' of natural habitat in 'seas' of human-altered habitats. The lesson of Island Biogeographic Theory was that these islands of preserved habitat could not maintain viable populations of many important species over time (Chadwick 1991). Island Biogeographic Theory has been refined by theories concerning meta-populations (Hanski & Gilpin 1991). This approach is more nuanced in that it considers sub-populations of species that are connected, to greater or lesser degrees, by exchange of individual organisms and propagules. The lesson of meta-population dynamics for landscape ecologists and biodiversity scientists alike is that neighbouring populations of the same species interact, often mutually increasing their probability of persistence, and therefore should not be managed in isolation. Thus a concern for 'connectivity' in the planning of conservation networks has arisen in which the connections between reserves, often referred to as conservation corridors, are seen as a means by which the negative effects of fragmentation and isolation can be reduced or overcome (Bennett 1999; Forman 1995; Guevara 1995).

Although the application of landscape ecology to conservation biology broadens the focus of conservation to include the landscape elements that connect areas, it does so under constraint. The constraint is that the landscape element of principal interest remains the habitat that has been determined to be most important for biodiversity conservation, often the least anthropogenically altered/most natural habitat. In Oaxaca this is usually the mature, naturally regenerated forest, where more species and more rare species are expected to be found, an expectation that is only rarely tested by assessment. Other landscape elements are of interest to biodiversity scientists only in relation to the degree to which they connect or isolate the forest and in how they might be manipulated to increase connectivity of that forest (e.g. Carrol & Kane 1999; Debinski & Holt 2000). The concern of Oaxaca's NGOs for all elements in a managed landscape that underpins the reconfiguration of biodiversity as a landscape phenomenon is subtly but crucially different. It implies an equality of importance between forest and other land uses, typically agricultural land uses. Hence, diversity between landscape elements has become the focus for Oaxaca's NGOs at the expense of diversity within a particular land use element. Thus the landscape reconfiguration proposed by these NGOs is not simply a practical application of landscape ecology.

In fact the reconfiguration proposed has more in common with the branch of applied biology that is agro-ecology with its concern for agricultural diversity. Certain land uses, other than mature native forest, have been shown to have the potential to play a role in biodiversity conservation independently of their relation to other landscape elements, and agro-ecosystems have for sometime been seen as a potential resource of biodiversity conservation (Brookfield & Padoch 1994; Pagiola *et al.* 1997). Examples from Mesoamerica include coffee plantations in Mexico (Moguel & Toledo 1999), secondary forest in Costa Rica (Finegan 1992) and kitchen gardens in Belize (Steinberg 1998). The reconfiguration of biodiversity as a landscape phenomenon is conceptually more in keeping with this tradition, particularly with its association with the movement towards sustainable agriculture (Vandermeer & Perfecto 1997). It is therefore proposed here that should Oaxacan NGOs require a scientific legitimisation of their approach to conservation, for example when dealing with funding partners, agro-ecology may be the place to look.

Accommodating the global and the local

Despite this potential for 'legitimisation' of biodiversity as a phenomenon of agricultural landscapes, the aim of Oaxaca's conservation NGOs is not the translation of one scientific regime of practice to another (compositionalism to functionalism, or landscape ecology to agro-ecology). They are motivated by the need to find accommodation between the global demand to conserve the Earth's biological variability and the local need to maintain and improve rural livelihoods. The translation is more about politics than it is about science and is an attempt to make the politics of biodiversity conservation workable for a local agenda. The landscapes of Oaxaca are not merely things amenable to technical/scientific management, but socio-political processes in themselves, 'with the power to produce relationships between people' (Morin 2003 p. 321). By reconfiguring, or translating, biodiversity into a landscape phenomenon, Oaxaca's NGOs hope simultaneously to maintain relationships with rural communities and with their national and international funding partners. Inevitably, whilst the origins of this translation are recognisable, there is also a loss (Wright 2002). The loss here is the decentring of the species as the 'fundamental unit of biodiversity' (Gaston 1996b p. 77) so that the need to assess species diversity becomes largely redundant as do recent theoretical advances in reserve site selection based on species assessments (e.g. Briers 2001; Margules *et al.* 1988; Pressey *et al.* 1997). Some of the NGOs active in Oaxacan conservation have only a rudimentary understanding of the fauna and flora that they are engaged in conserving. Hence biodiversity scientists are likely to lament the same lack of systematic conservation planning in Oaxaca just as elsewhere (Ledig 1988; Pressey *et al.* 1993). However such planning to a large extent reflects the difference between what to conserve, about which scientists know a great deal, and how to conserve it, about which there is less agreement (Bawa *et al.* 2004).

In Oaxaca local NGOs are expected to make good this difference in understanding and for them the reconfiguration of biodiversity is the first step in doing this. This has the potential to create problems with partners, such as WWF, that act as the transmission channels for biodiversity conservation (Townsend *et al.* 2002 p. 830) and that value scientific prescriptions for its enactment. Isabella of WWF explained one manifestation of the problem thus:

> When we have got to have support to justify a zone [for conservation] with real, true scientific information there are lots of errors...we do not have valid information to send because of the lack of training of the NGO technical staff.

However, this is tempered by recognition from within WWF that the discourse of biodiversity conservation may not entirely capture the complexities and challenges faced by its local NGO partners. Sally in WWF's Mexico City office offered this opinion:

> I don't like the concept that biodiversity is the be all and end all of conservation. More than anything because if you look at very often what creates diversity in eco-systems, it could be varying degrees of intervention, and you know the more diverse ecosystem is very often one that's got edge [is fragmented].

This allows for the possibility that the accommodation local NGOs have found between biodiversity conservation and rural development could also work as a negotiating position between local and funding partners.

8.7 Geographic Information Systems and Techno-Scientific Authority

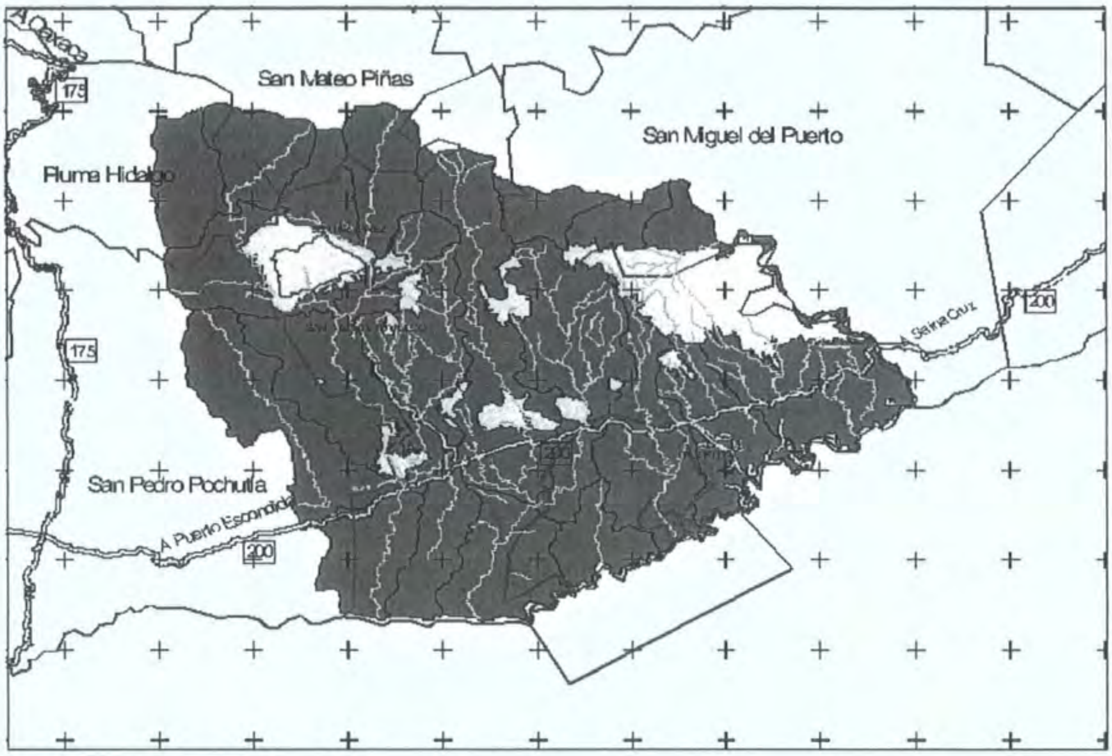
The decentring of the species as the basic unit of biodiversity and the lack of biodiversity assessment does not imply a rejection of all scientific discourses and practices on the part of Oaxaca's NGOs. Indeed the rhetorical advantage of biodiversity as a landscape phenomenon is that it continues to allow them to claim scientific authority for their activities. The desire to make such a claim is illustrated by their up-take of geographic information systems (GIS), all these local NGOs have some current or previous capacity in this techno-scientific practice. This is despite the considerable resource demands of its establishment and maintenance.

GIS is very well suited to this reconfiguration. Biodiversity as number of species lends itself to numeric forms of analysis and presentation; indeed it is the opportunity to quantify nature that explains much of biodiversity's appeal to the scientific community. In contrast, biodiversity as landscape offers the possibility of more visually mediated forms of presentation; landscape is after all, in the western tradition, a visual phenomenon. Compared to the complexity of the mathematics of species sampling and estimating total species numbers (e.g. Chazdon *et al.* 1998; Colwell & Coddington 1994; Gotelli & Colwell 2001; Plotkin *et al.* 2000), visual representations of landscape appear more immediately tractable, even if closer examination reveals a complex interaction of the social, the material and the symbolic lies hidden beneath (Nash 1999). The appeal of visual presentation is realised, and lent techno-scientific validity, through the practice of GIS.

It is difficult to ascertain the extent to which analysis by GIS has been used to inform the decision-making processes of Oaxaca's conservation NGOs. Here it is suggested that another role for these technologies may be at least, if not more, important. GIS is a collection of sophisticated technologies that allow the presentation of locally specific knowledge in a visually impressive manner. There is a power and aura to GIS (Dunn *et al.* 1997) that enables an NGO to demonstrate its technical competence and stake a claim to superior local knowledge.

GIS technologies require quantitative inputs, but much of the numeric data needed is relatively accessible in Oaxaca from the internet and from the state office of the National Institute of Statistics and Geography (INEGI) but the outputs of this quantification are visual and superficially simple. The same cannot be said of data capture for species diversity assessment as this requires far more logistical and specialist support, particularly for species identification, the latter capacity often only being available in the metropolitan centres of Mexico. The Oaxacan NGO that is most orientated towards the biological research, *Arbol Verde*, required over a decade, from the initiation of collections to the publication of a checklist, to complete a floristic survey of one small area of the state. This was due to the logistical constraints of plant collecting and the need to coordinate with the National Herbarium in Mexico City. In the same period the same NGOs has used GIS technologies to produce a series of cartographic representations of forest cover across the entire state.

Figure 8.1 GIS Respresentation of Santa María Huatulco showing Huatulco National Park (red outline), areas protected by under community/municipal agreement (pink), and unprotected forest and farmland (green).



The preference for GIS mediated interpretations of biodiversity does not imply that it is any more tractable to client groups in rural areas, Carlos of *Ecología* is aware of this:

> The question of maps, and above all geographic information systems is that it is a modern technological approximation that sometimes encounters a lot of local resistance, what happens is that these models are things that do not say anything to the people.

However, this may be irrelevant; the importance of GIS to local NGOs is that it is a relatively easy way to demonstrate scientific and technical sophistication to funding partners.

The uptake of GIS amongst Oaxaca's NGOs shows that their reconfiguring of biodiversity is not a rejection of the role of science in Oaxacan conservation *per se*,. Oaxaca's NGOs are willing to engage in scientific practice when they consider it useful to them, as is further confirmed by the spread of small NGO-supported herbaria in Oaxaca, as and when they see its benefits. However, there may be a cost even in this selective use of science.

In creating representations, practices such as GIS detach objects from their environments and create hybrid products of the social and the scientific, products which are both dependent on, and the result of, multiple translations in complex networks (Latour 1993). This is highly political; in capturing place based knowledge in this way Oaxacan NGOs become central in a process similar to that describe by Bryant (2002) in the Philippines whereby:

> Hitherto 'peripheral' people and biota are brought within a remit of political rationalities of control and surveillance.

The danger of GIS practices is that places are represented and hence mobilised in scientific networks without the human inhabitants of those places, on behalf whom the local NGOs claim to act, retaining control of them, a concern that may explain the criticism voiced by Carlos (above).

The outputs of GIS are not for the consumption of client groups, but for representing landscapes to distant actors, particularly funding partners. To borrow from Bender (1993 p. 1) the landscapes in question are 'close-grained, worked upon, lived in places' to rural Oaxacans, but 'distant and half-fantasised' when reworked and re-presented by the NGOs to their national and international partners. The transformation of biodiversity into a landscape phenomenon allows them to represent Oaxacan nature as a substrate for rural development whilst simultaneously maintaining it as scientific construct for funding partners interested in biodiversity conservation. GIS, in these circumstances, becomes a practice of techno-scientific justification for Oaxaca's NGOs. The result is that the international politics of biodiversity conservation funding can be articulated with the local politics of land use. Each of the images produced by GIS is a claim to know something about a place, and is therefore a 'powerplay' (Wright 2002 p. 414) in the politics of Oaxacan conservation. Every area delimited on a digitised map says to the world that the NGO is technologically sophisticated and scientifically literate, no matter that species within each area remain unknown. GIS gives them the means to do this, but also distances them from their client groups, thus local NGO-international NGO relations in Oaxaca follow a pattern suggested by Farrington and Bebbington (1993 p. 177) for local NGO-state relations in that:

> While they generally profess a closer affinity to the poor than to the state, they bear more resemblance to the state than to the poor - and in most of their activities they operate in a manner that is more akin to the state than to any organisation of the poor.

WWF has invested in the development of GIS amongst Oaxaca's NGOs, but with questionable direct influence on conservation as practiced in rural Oaxaca. Perhaps though this misses the point that GIS strengthens the partnership between local NGO and donor by lending techno-scientific authority to the reconfiguration of biodiversity as landscape.

8.8 Discussion

The reconfiguration of biodiversity as landscape, like any other translation 'consists of combining two hitherto different interests...to form a single composite goal' (Latour 1999 p. 88). The translation from a compositionalist/biodiversity approach to a functionalist/landscape approach maintains a concern for biodiversity conservation within a framework of rural development. It has been argued here that this is a strategic ploy in that allows the pursuit of rural development activities to proceed whilst maintaining the rhetorical advantages and funding options associated with the agenda and discourse of biodiversity conservation. Oaxaca's local NGOs consolidate this strategic advantage not through the systematic conservation planning promoted by conservation biologists (Pressey *et al.* 1993; Pressey *et al.* 1996) but through a different techno-scientific practice, GIS, and in so doing reach an accommodation with their funding partners whose primary concern is for biodiversity conservation rather than rural development.

Escobar (1999) has shown that nature is differently experienced according to social group and historical period but also that the relationship between different regimes of nature is not linear; one does not simply replace the other. In Oaxaca there is still a role for biodiversity conceptualised as a biology of numbers of species and for the emphasis this places on forests (e.g. Gordon *et al.* 2004; Salas-Morales *et al.* 2003). Through different practices the biodiversity assessor and the Oaxacan NGO call forth different representations of nature from the landscape whose importance lies not so much in the 'how' of the practice as the 'why' (Rose 2002). What Oaxaca's NGOs have done is produce a regime better suited to the negotiated position they wish to adopt and in doing this they have subverted the politics of biodiversity conservation. Just as regimes of nature do not follow a linear succession, the local enactment of biodiversity conservation in Oaxaca does not follow the predominant 'linear model of policy influence which assumes a one-way flow of information from science to policy and society' (Eden 1998 p. 426), indeed, it could be said that Oaxaca's NGOs are a site of resistance to it. However, this appears to be very much a tolerated resistance, at least on part of those donor partners with agendas for biodiversity conservation, as it has not presented a significant obstacle to continued collaboration.

Guha (1989) argues that environmental issues framed in the North are ultimately imperialistic when translated to the South and suggests that

> Green missionaries maybe more dangerous, and certainly more hypocritical than their economic or religious counterparts (Guha 2000 p. 369).

Esteva and Prakash (1998) reject the whole idea of understanding the global as impossible; nobody has an adequate perspective on the multiple 'pluriverses' that are spread across places and cultures. To them the global is given false legitimacy by statistical and scientific reductionism that leads to, among other things, the dangerous and destructive interventions of environmentalism. It is therefore contended that ethical development requires the destruction of the monopoly that scientific and technical rationalities have over environmental discourses (Goulet 1993). However, rather than destroying these rationalities, what some of Oaxaca's NGOs appear to be doing is adapting, translating and reconfiguring one of them, biodiversity conservation, to the situation they encounter in Oaxaca claiming scientific legitimacy as a strategic ploy in negotiating relations with their funders.

There is no single type of relationship, whether collaborative or conflictive, in Oaxacan conservation, but multiple and diverse interactions between various actors. Either partnership or resistance, or both, might characterise the various relationships each local NGO has with funding partners, biodiversity scientists or even their fellow NGOs. Hudock (2000 p. 17) argues that partnership is a poor descriptor of donor-NGO relations as many interactions are isolated incidents, rather than part of a larger process that partnership would imply. Certainly any relationship in which funds flow from one partner to another can be described in terms of inequality and Lister (2000 p. 229) suggests that control over money is 'the most frequently cited constraint to the formation of authentic partnerships'. However Oaxacan NGOs have considerable advantages that counterbalance this. Firstly, they have proven adept at diversifying their funding sources, none of the NGOs discussed are dependent on a single funding partner. Secondly, their very location close to Oaxaca's biodiverse habitats gives them considerable bargaining power; they can claim to know the sites and communities in question better than their partners do. This supports Lister's (2000 p. 236) contention that:

> It is not sufficient just to consider asymmetries of power between agencies as constraints to partnership, but the wider framework within which those agencies operate, and the mechanisms for establishing those frameworks including the use of discourse, must also be taken into consideration.

The rhetoric of partnership, to which the national and international funders of biodiversity conservation subscribe, demands that working relationships with local organisations are formed despite subtle differences in the constructions of nature to which different partners ascribe. Funding partners, such as FMCN and WWF, are content that biodiversity conservation will happen regardless of their local partner

NGOs' differing views of what biodiversity might mean. Consequently, the prescriptions of biodiversity scientists cannot be channelled directly, without modification, to Oaxaca's endangered habitats. Thus the criticism of Sundberg (1998) that scientific and technical expertise has been used to detach protection of the Maya Biosphere Reserve in Guatemala from the socio-political realities in which it is embedded cannot is not applicable in Oaxaca. Biodiversity as an organising concept provides sufficient flexibility in Oaxaca to accommodate both local agendas and a modification of the biodiversity agenda.

As a result of the reconfiguration of biodiversity as a landscape phenomenon of pastures, forests and home gardens, the domains of nature and culture are no longer managed separately and so the NGOs

> Debunk the nature/culture dichotomy that is fundamental to the dominance of expert knowledge' (Escobar 1999 p. 9).

In Oaxaca the *ex cathedra* pronouncements of environmental experts (Norton & Hannon 1997) are contested, at least where local NGOs mediate. For better or for worse, biodiversity assessment, which would otherwise give the expert grounds on which to pronounce, is not seen as a priority by these NGOs.

It is therefore argued here that the NGO mediated north-south partnership described represents an advance over prescriptive models of the implementation of the international biodiversity conservation agenda. Oaxaca's NGO community adapts concerns for biodiversity conservation to their more local perspective. Farrington and Bebbington (1993) summarise the variability of NGO-state relations as being somewhere on a continuum between confrontational to productively synergistic. If this is extended relations between non-governmental donors and local NGOs, then in Oaxaca these relationships, taken as a whole, are nearer the synergistic pole.

However, the caveat is that it cannot be assumed that the voice of local NGOs is that of the rural communities who are likely to be most affected by conservation interventions. Oaxaca's NGO community is primarily composed of middle class university graduates, many of whom are not Oaxacan by birth. These characteristics undoubtedly make them attractive to funding partners; they share an understanding, if not necessarily an interpretation, of the politics and idiom of biodiversity conservation. Whilst it is undoubtedly true that these NGOs are deeply embedded in

Oaxaca and interact intensively, if not uniformly, with their rural client groups, the cultural distance between these NGOs and Oaxacan *campesino* communities is great. It requires an act of faith to assume that *campesino* voices are faithfully interpreted, relayed and acted upon by the NGO community. For their international and national funding partners this is not sufficient grounds for prohibiting them from advancing biodiversity conservation through local NGO mediation. For *campesino* communities, Oaxacan NGO may well appear as just another external actor interested in imposing a regime of practice on their landscape.

Chapter 9: An assessment of *ad hoc* reserve selection using systematic reserve selection criteria: a case study from Oaxaca's dry forests

9.1 Introduction

Protected areas, or reserves, remain the most important tool for the conservation of biodiversity, with over 10% of the Earth's terrestrial surface now being included within some sort of biological reserve (Chape *et al.* 2003). Whilst reserves fulfil many roles, the conservation of biodiversity is considered to be one of the primary reasons for their establishment. This is supported by international agreements such as the Convention on Biological Diversity's Plant Conservation Strategy (CBD 2002) which sets a target of 60 per cent of the world's threatened plant species to be conserved *in situ* by 2010. Expansion of the global protected area network has been accompanied by increasing interest in how reserve sites are selected and debate about the deficiencies of *ad hoc* selection (Pressey 1994; Pressey *et al.* 1993). *Ad hoc* selection in this context means the selection of reserve sites for reasons other than that of maximising the representation of biodiversity. Such selection has resulted in many reserves being sited in areas under little adverse pressure from human populations and consequently reserve systems typically under-represent habitats and biota from biomes that are most threatened and where the opportunity cost of protection is greatest (Ledig 1988; Pimm & Lawton 1998). A related issue is that of reserve selection based on the presence of one or few charismatic species (Gaston & Rodrigues 2003). In either case it is widely argued that inefficiency results and Pressey (1994) identifies two related shortcomings of *ad hoc* reserve selection. The first is that some elements of biodiversity are likely to be left outside of an *ad hoc* reserve system. The second is that *ad hoc* selection can make the task of ensuring that all diversity is represented within a reserve system more expensive because more sites than the minimum necessary, or more expensive sites than necessary, will be selected.

Systematic reserve selection is defined in opposition to *ad hoc* selection (Margules & Pressey 2000; Pressey *et al.* 1993). It attempts to maximise the number of species represented in a reserve network subject to some form of resource constraint- typically that the total number or area of reserves should be minimised.

The principle of complementarity has a central role in systematic reserve design (Howard *et al.* 1998; Rodrigues & Gaston 2002; Vane-Wright *et al.* 1991) with sites being prioritised according to the extent to which their protection will add to, or complement, the list of total species in previously selected reserves. Therefore, the next most important site to add a system is not necessarily the one with the most species, but the one that adds most species not already present. Systematic selection therefore implies a far greater use of biological information than does *ad hoc* selection and that information usually takes the form of species lists from each site.

Several approaches to systematic selection have been adopted. These include GAP analysis (Scott *et al.* 1993) which usually uses remote sensing data to identify habitats under-represented in existing reserve networks, and the use of computerised algorithms to identify sites containing high numbers of species under-represented in reserve networks (Pressey & Cowling 2001). Here algorithmic selection is the central concern and therefore species diversity, rather than habitat diversity, the subject of consideration. Systematic reserve selection is now widely promoted (Margules & Pressey 2000; Pressey *et al.* 1993) and increasingly, although not commonly, implemented (e.g. Pressey *et al.* 2003).

The constraints that the total number or area of reserves should be minimised is a crude measure of efficiency given that the costs of establishment and management of reserves are neither likely to be equal for all sites nor to be in linear proportion to the area of each site. More sophisticated cost measures have therefore been introduced (e.g. Ando *et al.* 1998; Pence *et al.* 2003) to measure the cost of establishing reserves. In doing this they effectively increase the influence of socio-economic criteria on reserve selection, but in a manner that is explicit and whose effect on representativity can be gauged.

In narrowing the criteria for reserve selection to the maximisation of species representation, other concerns are ignored. These could include the viability of the reserve in the face of human threats, the scenic or touristic value of a landscape or the higher value society places on some charismatic or economically important species. Socio-economic criteria, such as the cost of acquisition, tenure, local legislative framework and local sympathy towards conservation are likely to continue to be determinants of reserve network design. Reserve designs that reflect these concerns are more likely to be socio-economically acceptable and therefore have a greater potential for long-term viability. Thus systematic reserve selection may have theoretical advantages, but the criteria that typically influence *ad hoc* selection might have practical benefits for the sustainability of reserve networks. Here, an *ad hoc* reserve network is tested using a data set from Mexican tropical dry forest to estimate the degree to which it falls short of the theoretical optimality, and the consequences of the gap between theory and practice are examined.

Tropical dry forests are considered to be amongst the world's most endangered terrestrial ecosystems (Lerdau *et al.* 1991) with those of Mesoamerica being of particular concern (Janzen 1988), hence a dry forest conservation initiative in Mexico makes a suitable case study for testing reserve selection efficiency. Compared to tropical rainforests, which have received much greater attention in scientific and popular discourse, tropical dry forests have been under-studied and are under-protected. Furthermore, the expansion of Mexico's system of nature reserves has been driven by factors other than the need to maximise biodiversity conservation (Cantú *et al.* 2004), this being as true for Mexico's dry forests as it is for any other of its biomes. Attention has begun to be directed towards the dry forest biome and recently more accurate quantitative assessments of its status have begun to emerge, both globally (Miles *et al.* in press) and within Mesoamerica (Gillespie *et al.* 2000; Trejo & Dirzo 2002). These assessments have confirmed that tropical dry forest is threatened, with intact dry forests in Mexico now occupying just 27% of the area that once would have supported this forest type (Trejo & Dirzo 2000). Areas subject to seasonal drought are thought to have been preferred by agriculturalists (Murphy & Lugo 1986) as drought suppresses pests and diseases, and enables the use of fire as a land management tool. Hence there has been a long association between the location of tropical dry forests and areas of human occupation, resulting in widespread forest loss and fragmentation (Maass 1995). However, human-dry forest interactions are not always negative and a rich ethnobiology has begun to be uncovered for these forests (Bye 1995) and the conservation of dry forest diversity based on positive human-forest synergies has begun to be explored (e.g. Boshier *et al.* 2004).

In this investigation the tree and shrub diversity of an *ad hoc* selected network of small, locally managed, dry forest reserves (the Communal System of Protected Areas- see below) is assessed for the degree to which the network includes the most diverse sites available. This is done by comparing rapid biodiversity assessments of the tree flora of each of the reserves with similar assessments from neighbouring unprotected forests. Thus a type of GAP analysis is performed in which, first, the potential of each of the currently unprotected forests to contribute to the overall list of protected species is assessed and, second, the potential of each of the currently unprotected forests to contribute to the overall list of protected *threatened* species is assessed. Selections made under each of these two scenarios are compared to a classification of the sites carried out by the ordination programme Two Way Indicator Species Analysis (TWINSPAN). Given that reserve selection is never based on complete inventories of all biota, the sensitivity of the selection procedures to variation in data quality is also considered here.

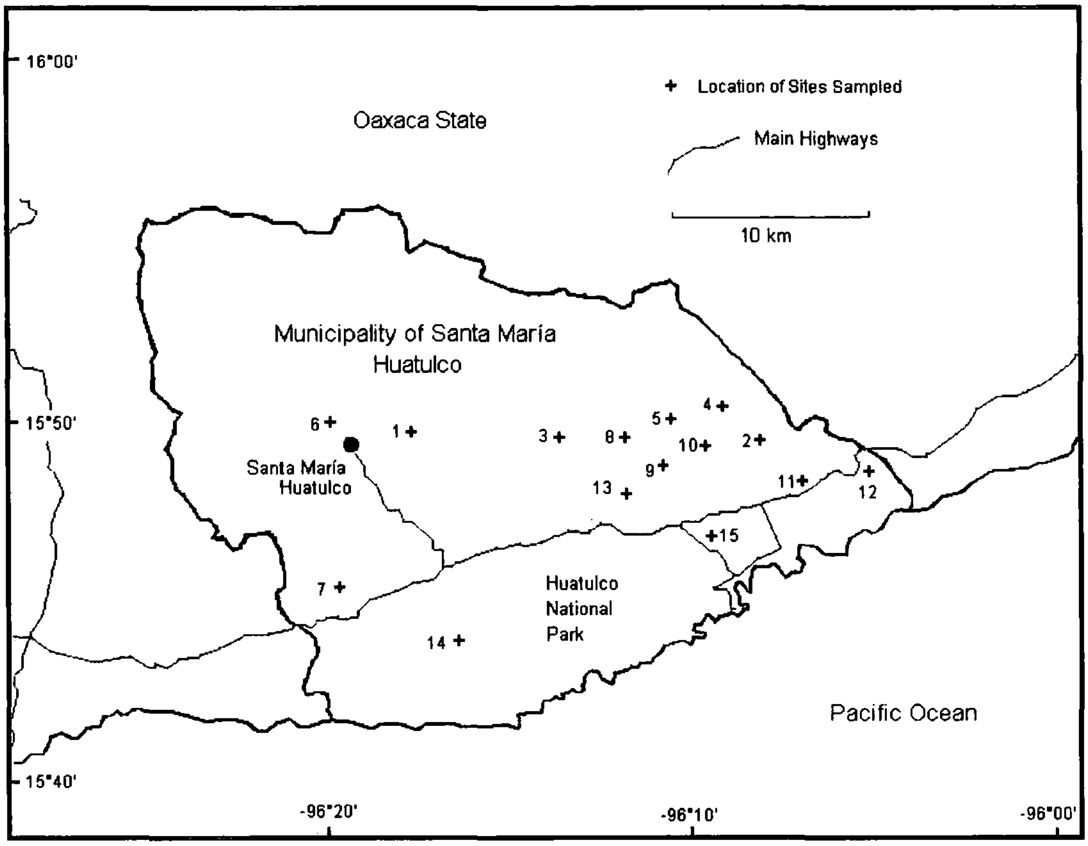
It has been argued that in an optimal conservation strategy every species should be represented more than once (Pimm & Lawton 1998) and therefore this network of small reserves is also assessed in relation to the species known to be found in the dry forest of a nearby federally managed national park. These two conservation initiatives have different administrative and legal statuses and therefore together increase the chances that any species represented in both might survive future administrative changes. This assessment considers the degree to which the network of locally managed reserves duplicates the species conserved in the national park, as well as complements them by adding additional species to the total listed of protected species.

9.2 Methodology

9.2.1 Study Site

The municipality of Santa María Huatulco in Oaxaca Southern Mexico (Fig 9.1) contains areas of deciduous dry forest within a matrix of agricultural land and, in the foothills towards the north, semi-deciduous forest. Most of the land in the municipality is held under communal tenure by the municipal authorities. Within the forest-farm matrix, 14 areas of forest in a variety of successional states of have been designated as natural reserves by the municipal and communal authorities. These areas contain two forest types corresponding to the 'low deciduous forest' and 'medium semi-deciduous forest' of Rzedowski (1981). The former is taken to be synonymous with tropical dry forest and is the unique focus here.

Figure 9.1 The Municipality of Santa María Huatulco, Oaxaca, Mexico



The reasons for the reservation of each area vary but include watershed protection, soil conservation and potential for future sustainable management. The selection process was not informed by biodiversity assessment, a prerequisite of systematic selection, and therefore would be considered *ad hoc* by the definition of Pressey (1994). With the aid of a locally based non-government organisation, these reserves have been officially designated as the Communal System of Protected Areas (hereafter SCAP, its Spanish acronym) to be maintained under forest cover indefinitely. The sites fall within an area of approximately 300 km^2 with altitudes varying between 20 and 150 m.a.s.l. From amongst the reserves seven areas were considered to be of tropical dry forest and of a successional status sufficient for a closed canopy to have formed. These seven sites were sampled along with eight alternate unprotected sites within the municipality that, at least by their biophysical conditions, could be considered potential reserves and that were accessible to a survey team. The forest patches sampled varied considerably in size from under 50 ha to over 300 ha.

From within the 6 000 ha Huatulco National Park four accessible sites considered to be of close canopy dry forest were selected randomly and surveyed. A composite species list was then compiled for this park from these additional samples plus a previously compiled, unpublished checklist.

9.2.2 Survey Method

The recognition that much planning for biodiversity conservation is based on inadequate information has lead to the development of rapid biodiversity assessment techniques (Phillips *et al.* 2003; Stern 1998) and it is a rapid survey methodology that is used here. The fifteen forested sites were surveyed using a fixed count methodology similar to that of Hall (1991) in which plots consisted of the 15 trees ≥ 5 cm diameter closest to a randomly located point. Following the recommendation of Condit (1998) that fixed count methods for tree diversity surveys should have a minimum of 100 stems per sample, each forest reserve was surveyed using 15 of these plots resulting in 225 individual trees surveyed per site. Plots within each site were located randomly, with respect to a 'base line' formed by the trail followed in order to enter the forest. The majority of trees encountered were identified in the forest. For those which were not identifiable, or whose identity was considered in anyway doubtful, voucher specimens were taken and deposited in the QUIE community herbarium in Oaxaca for later identification with the aid of expertise from the National Herbarium (MEXU) in Mexico City. Vouchers that could not be confidently identified to species were treated as morphospecies (Kerr *et al.* 2000) in the analysis below.

9.2.3 Analysis

For this analysis the assumption is made that each site surveyed, either in the SCAP or amongst the unprotected sites, would be equally costly to conserve, regardless of size, location, content or history.

The total number of species observed, $S_{(obs)}$, for the complete survey was calculated and compared to two non-parametric estimators of diversity. In a comparative study Chazdon *et al.* (1998) showed that, of a range of non-parametric estimators, the 'Incidence Coverage Indicator' (ICE) and 'Chao 2' satisfied the requirements of a species-richness estimator for a Costa Rican tropical forest. Therefore, these two estimators are used here. For ICE there is as yet no known way of calculating the variance of the estimates and therefore confidence limits could not be calculated.

TWINSPAN was used to provide a classification of the surveyed sites based on the presence/absence of observed species (McCune & Mefford 1997). The efficiency of the SCAP was then measured as the percentage of forest types identified by TWINSPAN that were represented in the SCAP. Redundancy in the SCAP was calculated as the percentage of sites that could be eliminated from the SCAP whilst maintaining current representation of forest types.

The sites were then compared using the three diversity measures; species observed $S_{(obs)}$, Fisher's α and Simpson's index (Magurran 1988) calculated by EstimateS 7.00 (Colwell 2004b). Redundancy within the SCAP was calculated as the number of its sites required to ensure representation of all species observed within it, measured as a percentage of the number of sites (seven) conserved. The efficiency of the SCAP in representing all species surveyed in the municipality was calculated as the percentage of all species in the 15 sites surveyed that were also found in the seven sites of the SCAP. The most efficient reserve design from amongst the 15 sites surveyed was calculated using the Greedy algorithm (Briers 2001), run as a freeware macro (http://users.aber.ac.uk/rob/reserves) in the spreadsheet programme Excel™, to show how closely the *ad hoc* SCAP approximated to a systematically selected reserve system. The Greedy algorithm was chosen because of its simplicity and long history (Margules *et al.* 1988) and also because many other heuristic algorithms for site prioritization that are now in use are variants of this basic complementarity algorithm (Sarkar forthcoming). It selects the most speciose site first, then the site that contains most species not already represented in the first sample, followed by the site with most species not already represented in the first two sites, and so on until 100% representation is achieved. The preceding analysis was then repeated using only those species determined to be threatened because of their restricted global ranges.

Finally, the SCAP was tested with respect to the neighbouring Huatulco National Park. The degree to which species in the National Park were also represented in the SCAP was used as a measure of the potential of the SCAP to provide supplementary conservation for the species of the National Park. The number of additional species the SCAP added to those of the National Park was also calculated to measure the potential of the SCAP to complement the National Park and hence the most important sites in the SCAP for complementing the National Park were identified using the Greedy algorithm. Thus, a type of GAP analysis was performed in which the elements of the SCAP needed to fill gaps in the species coverage of the National Park were identified.

Threatened species

Estimates of the range size of each species were based on herbarium specimen information held in the National Herbarium in Mexico City and in the *Tropicos* database of the Missouri Botanic Gardens (w^3Tropicos undated). Given the variable precision with which herbarium specimen localities are recorded, the number of Mexican states/Central American countries in which a species had been collected was used to estimate distributions. The first step in creating a weighting was to calculate a relative area for each state and country by dividing the land area of each of those political entities by the land area of Oaxaca (thus giving Oaxaca a relative area of 1). To create a score for each species, the total relative land area of the states/countries in which each species was found were then calculated and inverted. Thus, the highest score obtained for any species was 1 (for Oaxacan endemics) with all other species, which must be known from Oaxaca and at least one other state/country, scoring greater than 0 and less than 1. In this way the species of most restricted range could be identified as those with the highest scores. Gordon *et al.* (2004) classified approximately 20% of the Mesoamerican dry forest species they identified in this region as being of conservation concern. Hence in the analysis that follows the most restricted 20% of species were taken to be threatened. Restricted range is used here as a surrogate of extinction risk, that is, the assumption is made that species occupying less of the earth's surface are more likely to go extinct than those that occupy a greater area.

9.3 Results

Figure 9.2 Species accumulation curve and the incidence coverage estimator (ICE) of total species richness for rapid botanical surveys of tree diversity in 15 dry forest sites in Oaxaca, Mexico.

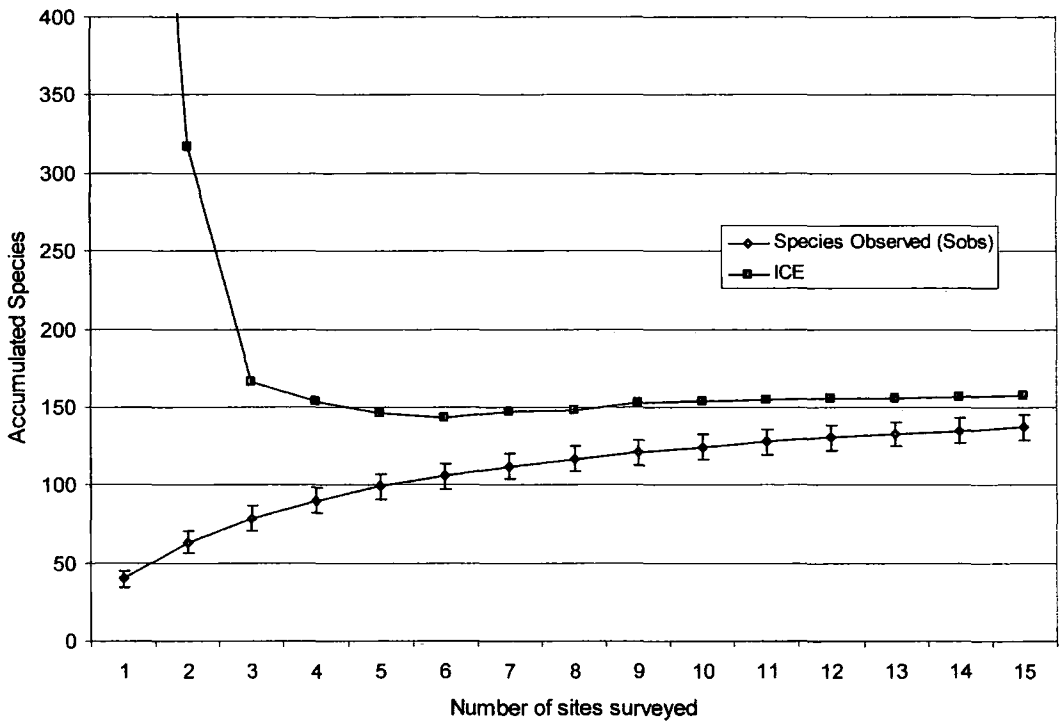
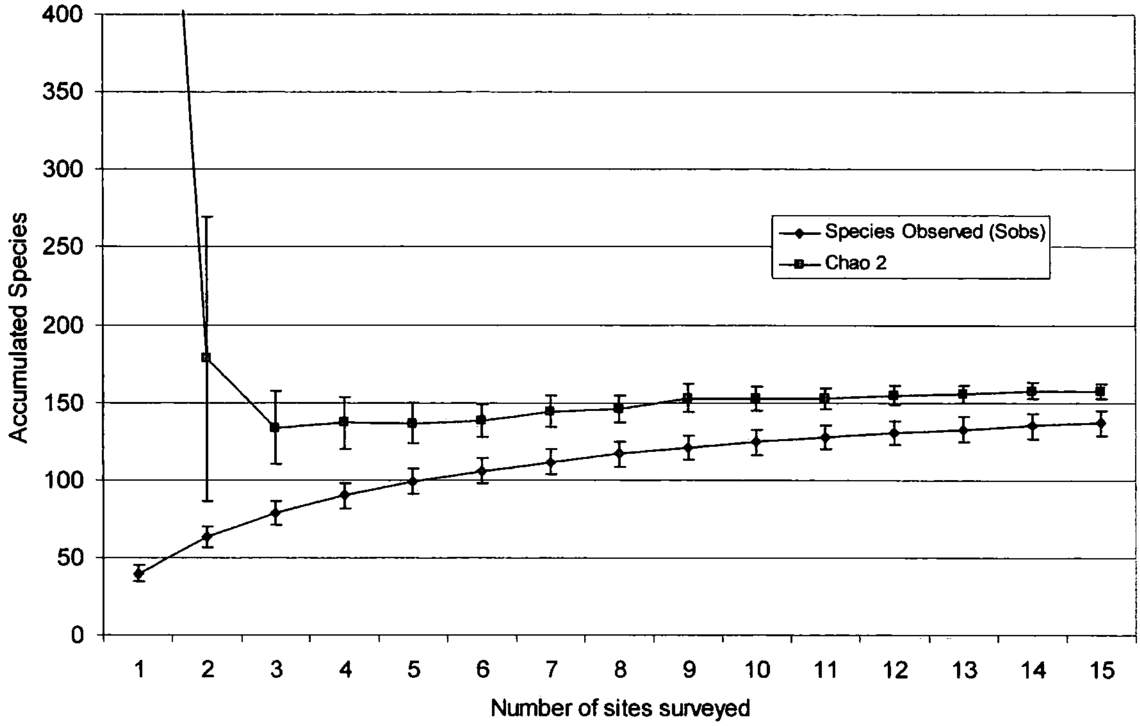
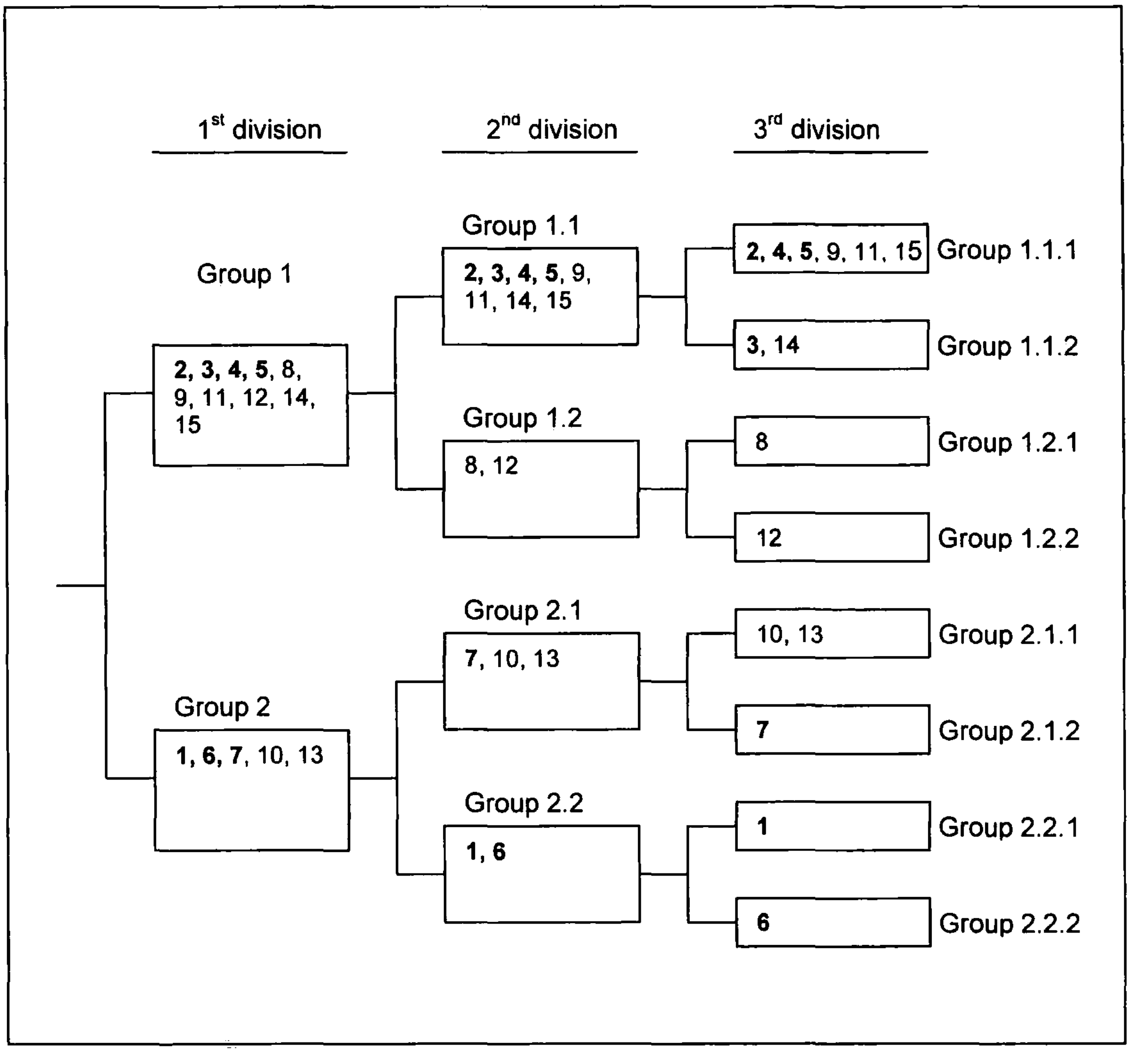


Figure 9.3 Species accumulation curve and the Chao 2 estimator of total species richness for rapid botanical surveys of tree diversity in 15 dry forest sites in Oaxaca, Mexico



The total number of species observed for the 15 sites sampled was 137. The species observed [S(obs)] accumulation curve in figures 9.2 and 9.3 has not reached an asymptote suggesting that there are more species to be found in this landscape, thus the following analysis, in common with other selection exercises, is based on imperfect information. The non-parametric estimators, ICE and Chao 2, both appear to be nearer to asymptote than S(obs). The estimators predict similar total species of 157.2 (ICE) and 157.5 (Chao 2). There is no overlap of the 95% confidence limits and therefore the estimated vales are statistically different from S(obs). This suggests that a selection exercise using these samples would be based on estimates of the distribution of approximately 87% of the total species in the landscape.

Figure 9.4. TWINSPAN analysis of tree diversity of fifteen dry forests sites in Oaxaca, Southern Mexico. Sites 1 - 7 (bold) correspond to the SCAP, sites 8 - 15 are unprotected.



In figure 9.4 reserves from the SCAP are represented in both groups 1 and 2 demarcated by the 1st division of TWINSPAN, there is therefore 100% representation but 71% redundancy as only two of seven sites are required for this degree of representation. However, SCAP sites are not represented in group 1.2 and therefore the SCAP is only 75% efficient in representing the 2nd division forest types and, given that only 3 of the seven are required to do this, this indicates 57% redundancy. With only seven SCAP sites in the survey, all eight forest types identified by the 3rd division could not be fully represented, thus the five types which are represented (Groups 1.1.1, 1.1.2, 2.1.2, 2.2.1 and 2.2.2) equate to 71% representation with 29% redundancy.

Table 9.1 Diversity scores for tree species found in fifteen dry forests sites in Oaxaca, Southern Mexico.

SCAP sites	S1	S2	S3	S4	S5	S6	S7	
S(obs)	39	51	48	50	58	27	36	
Fisher's α	15.32	20.62	18.69	19.93	25.31	8.01	12.1	
Simpson's index	17.62	33.62	25.23	22.3	32.18	6.93	15.57	
Unprotected sites	S8	S9	S10	S11	S12	S13	S14	S15
S(obs)	42	49	33	52	11	38	36	24
Fisher's α	15.22	19.3	10.65	21.20	2.42	13.10	12.09	6.8
Simpson's index	15.2	21.03	13.65	15.13	4.41	13.29	13.31	11.14

Assuming that sites with the highest diversity should be accorded higher priority for conservation, the rank orders of importance for the fifteen sites suggested by the three diversity scores in table 1 are highly correlated (S(obs) v Fishers α, Spearman's rho = 0.996; S(obs) v Simpson's index, Spearman's rho = 0.829; Fishers α v Simpson's index, Spearman's rho = 0.843; all cases $p < 0.01$). This suggests that the computationally simplest, S(obs), is a useful indicator of diversity amongst the sites. Whilst SCAP samples have a higher mean S(obs) than the unprotected sites (44.1 to 35.6 respectively) the difference is not statistically significant (t-test $p > 0.05$) and the ranking of the sites by this metric suggest no significant difference between them (Mann-Whitney U: $p > 0.05$). If seven of the fifteen surveyed sites were selected according to the ranking suggested by these diversity scores, rather than by the *ad hoc* method that resulted in the seven sites of the SCAP, a different reserve network would have been chosen in each case (table 9.2). Using S(obs) to rank the sites would result in three of the current seven reserves being rejected; using Fisher's α rejection of two of the seven would result, and using the Simpson index suggests that four of the seven should be rejected. In each case Site 6 is rejected from the protected reserves and site 9 which is currently not protected in the SCAP is included.

Table 9.2 The order of priority for a seven-site reserve network selected from 15 dry forest sites in in Oaxaca, Southern Mexico by three selection criteria based on species diversity. The SCAP sites represent the extant reserve network and are not ordered by any variable. Sites chosen by the selection criteria are ordered most important (highest score) to lowest from left to right

	Site						
Ad hoc (SCAP sites)	S1	S2	S3	S4	S5	S6	S7
Selection criteria							
S(obs)	S5	S11	S2	S4	S9	S3	S8
Fisher's α	S5	S11	S2	S4	S9	S3	S1
Simpson's index	S2	S5	S3	S4	S9	S1	S7

Selecting the seven sites with greatest S(obs) would result in none of the Group 2 forest types being included in a reserve and therefore one of the two principal forest types identified in the municipality would go unprotected using this metric. The use of Fisher's α for ranking suggests a slight improvement in representation of species as both Groups 1 and 2 would be represented in a seven site reserve network based on ranking by this metric; however, Groups 1.2 and 2.1 of the second TWINSPAN division would not be (50% representation, 71% redundancy). Simpson's index gives slightly better coverage of the forest types identified; only Group 1.2 of the second division is missing (75% representation, 57% redundancy).

Complementarity

In order to achieve complete representation of species found in the seven sites within the SCAP, the Greedy algorithm selected all of the seven sites (table 9.3); therefore each survey site contains species unrepresented elsewhere in the network. By this measure there is no redundancy within the SCAP. To achieve the goal of representing every observed species in all of the fifteen sites surveyed, the algorithm selected eleven of the fifteen sites. Of those eleven sites, five are already protected within the SCAP. The two sites from the SCAP not selected by the algorithm, sites 3 and 7, are its 4^{th} and 6^{th} most diverse sites by S(obs). The sites selected by the algorithm follow a pattern that could be predicted by TWINSPAN (figure 9.4). The first two sites selected are from Groups 1 and 2, that is, they represent each of the two forest types defined by TWINSPAN's first division. The following four sites selected represent one of each of the four forest types of the second division defined by TWINSPAN, Groups 1.1, 1.2, 2.1 and 2.2. The remaining five sites selected are from four of the eight forest types identified by the third division of TWINSPAN. This suggests selecting forests to represent each of the forest types identified by TWINSPAN provides an approximation of a complementary reserve design. Note that the three final sites chosen, sites 10, 14 and 12, add just four species to the total represented.

The total S(obs) for the seven sites in the SCAP was 118. Out of a total for the fifteen surveyed sites, this gives an efficiency of 86%. If the seven most diverse sites by the S(obs) metric are selected to form a reserve network it would contain 108 species, an efficiency of 79% whilst the first seven sites chosen by the algorithm, when both protected and unprotected sites are included, contain a total of 130 species, an efficiency of 95%. If the SCAP is considered to be established and not renegotiable, then to achieve 100% representation of the species observed in all fifteen surveys a further six of the unprotected sites would have to be added to it. This corresponds to a GAP analysis of the SCAP. This total of thirteen sites is two more than required if the SCAP is considered negotiable. The contribution of the last selected sites is, again, very low with the final four sites adding just six new species between them.

Table 9.3 Reserve network selection for 15 sites in Oaxacan dry forest using the Greedy selection algorithm.

	Within-SCAP selection (7 sites only)		Sites selection to achieve 100% species representation		Sites added to SCAP to achieve 100% species representation.	
Order of selection	Site	Cumulative species	Site	Cumulative species	Site	Cumulative species
1	5	56	5	56	SCAP	118
2	1	81	1	81	8	126
3	2	93	11	97	11	131
4	7	103	8	109	13	133
5	6	111	6	118	10	135
6	4	115	13	125	14	136
7	3	118	2	130	12	137
8			4	133	9	Not selected
9			10	135	15	Not selected
10			14	136		
11			12	137		
12			3	Not selected		
13			7	Not selected		
14			9	Not selected		
15			15	Not selected		

The correlation between the order in which the diversity indices (table 9.2) selected sites and the order which the Greedy algorithm selected them (unselected sites ranked equal last) is positive, but not significant (algorithm v S(obs), Spearman's rho = 0.434; algorithm v Fisher's α, Spearman's rho = 0.434; algorithm v Simpson's index, Spearman's rho = 0.133, all cases $p > 0.05$).

Figure 9.5 Average number of individuals of each remaining species to be included in a reserve network following successive selections by the Greedy algorithm.

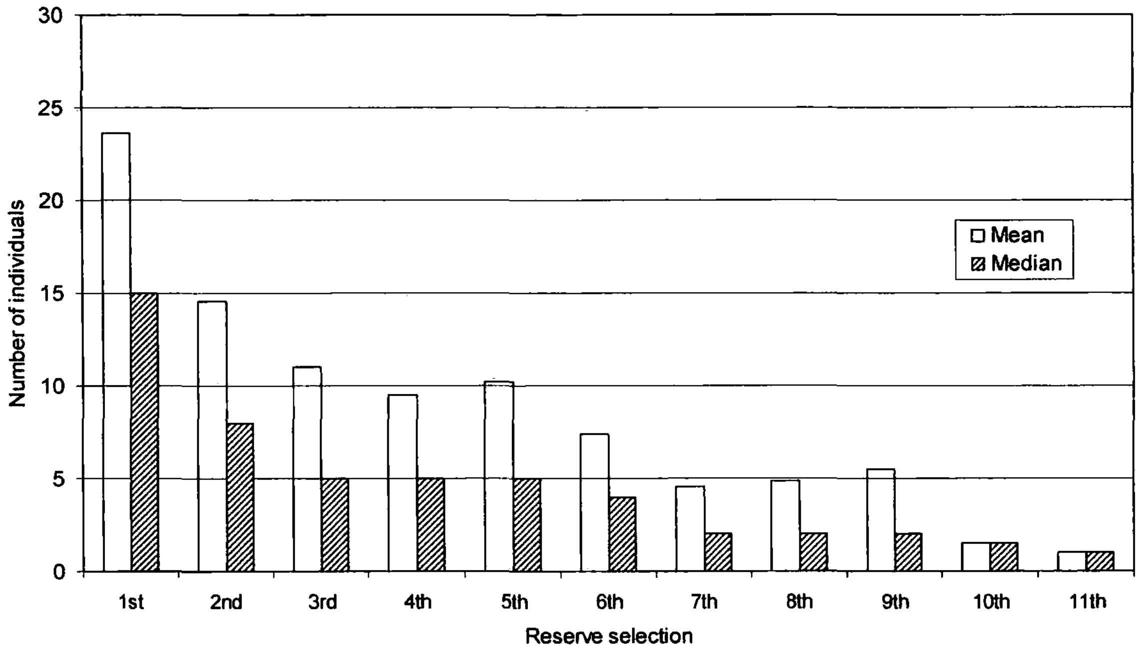


Figure 9.5 shows that the Greedy algorithm selects a disproportionately high number of the locally most common species in the initial stages of site selection, with later selections being amongst the locally less abundant. The surveys of these forests revealed that a few species are represented by many individuals and most by very few, hence the medians in figure 9.5 are smaller than the means. This species-abundance distribution is typical of a diverse naturally regenerating tropical forest (Hubbell 2001). The implication is that site selection is sensitive to species abundance, with a selection of later sites being dependent on the distribution of not only very few species but also very few individuals of those species. This is more than an artefact of sampling intensity. More intensive sampling would increase the number of individuals of each species, only for other even less locally abundant species to be captured in such surveys.

Threatened species

Figure 9.6 shows the percentage of restricted range species in the total species complement of the 11 sites chosen by the selection algorithm to efficiently represent the total S(obs) (see table 9.3). The restricted range species make up 20% of the total, but there appears to be no obvious pattern here, with some selections adding more than 20% and others less. The restricted range species therefore did not drive the site selection process for total species representation. To bias the selection towards representation of such species therefore necessitated unique consideration of these species.

Figure 9.6 Percentage of restricted range species amongst species added by successive reserve selections from amongst 15 dry forest sites in Oaxaca. Eleven sites chosen by the Greedy selection algorithm to represent all species.

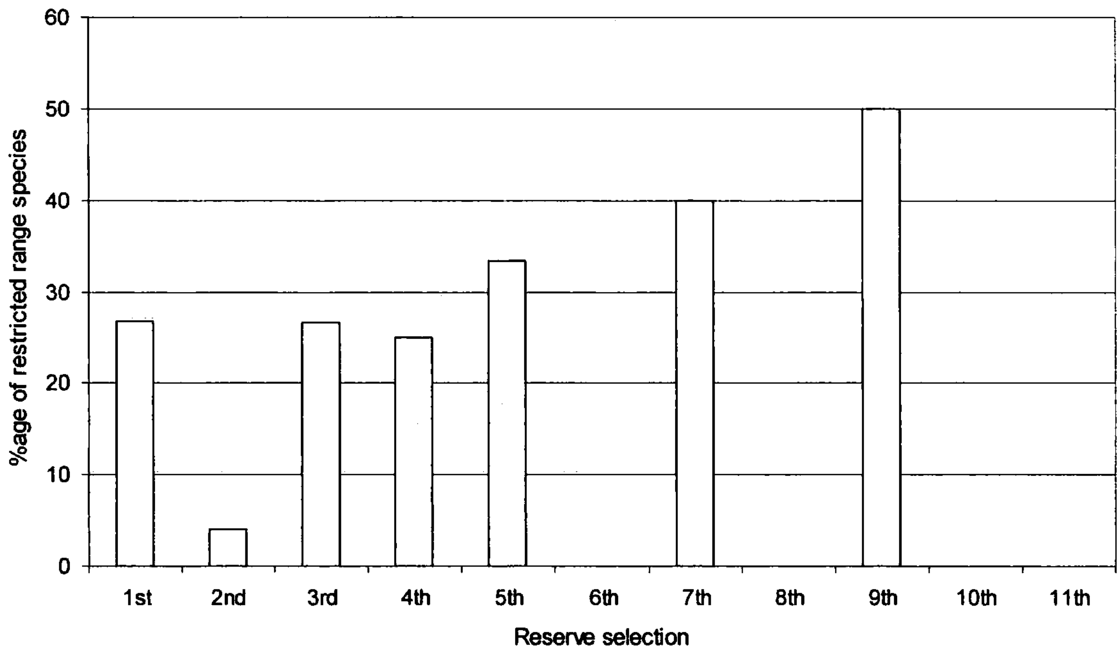
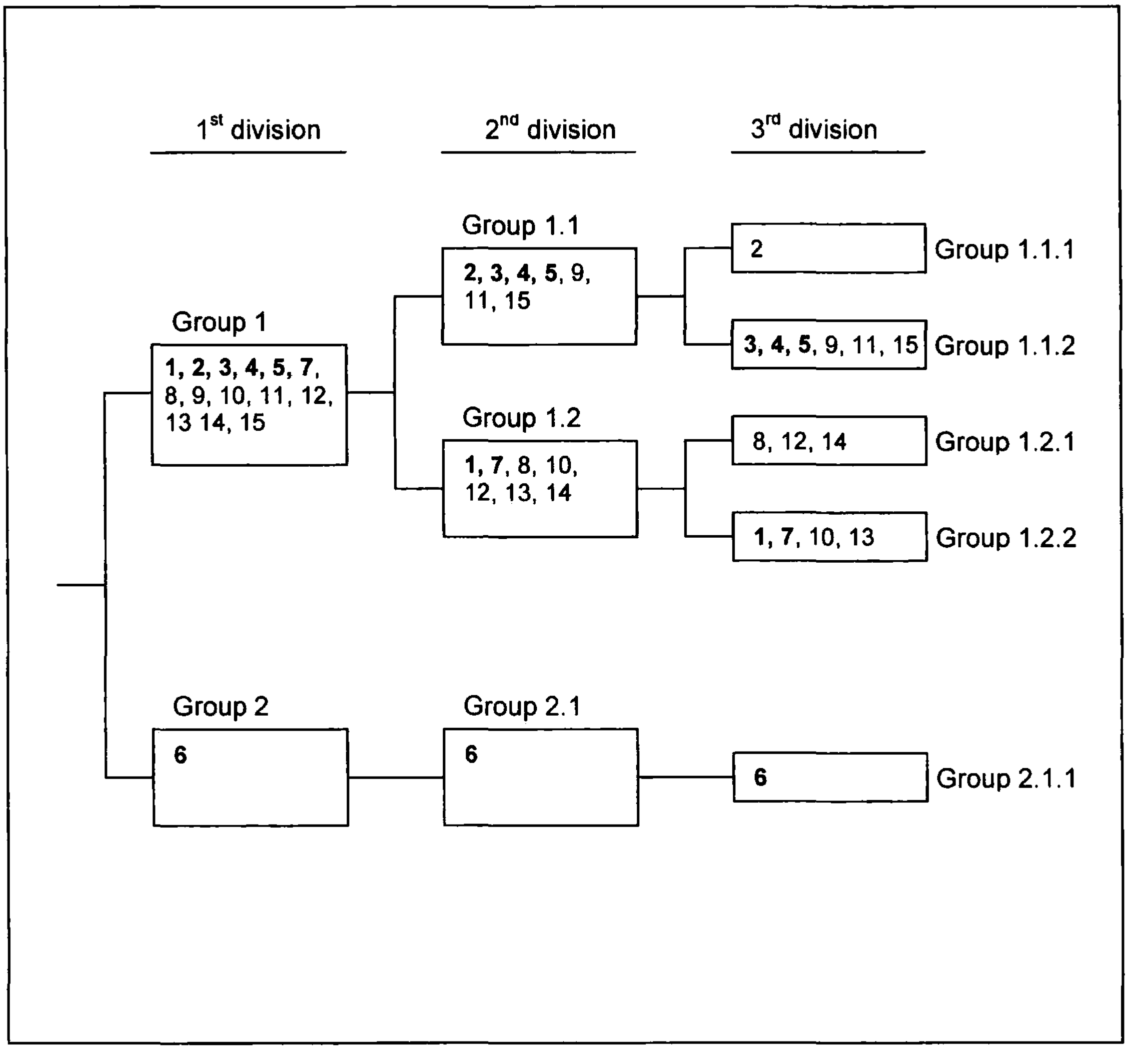


Table 9.4. Diversity scores for restricted range rare tree species found in fifteen dry forests sites in Oaxaca, Southern Mexico.

SCAP	S1	S2	S3	S4	S5	S6	S7	
S(obs)	2	12	9	12	15	4	6	
Fisher's α	0.85	4.19	4.07	4.83	5.38	1.16	2.21	
Simpson's index	2.15	8.17	6.36	6.17	7.12	2.97	2.46	
Unprotected	**S8**	**S9**	**S10**	**S11**	**S12**	**S13**	**S14**	**S15**
S(obs)	8	10	5	17	3	5	8	10
Fisher's α	4.20	4.42	1.52	6.96	1.28	7.82	4.52	2.72
Simpson's index	7.07	7.17	3.11	10.81	1.78	10.5	7.96	5.37

The rank orders of importance for the fifteen sites, based on their content of restricted range species and suggested by the three diversity indices in table 9.4, are positively correlated (S(obs) v Fishers α, Spearman's rho = 0. 686; S(obs) v Simpson's index, Spearman's rho = 0. 654; Fishers α v Simpson's index, Spearman's rho = 0.861, all cases $p < 0.01$). Although these correlations are weaker than similar comparisons using all species surveyed at each site, they are statistically significant and suggest that the computationally simplest, S(obs), is a robust indicator of restricted range tree diversity. Also, the rank order of sites by total S(obs) (table 9.1) is highly correlated with the rank order of sites by restricted range S(obs) (Spearman's rho = 0.751, $p < 0.001$). Whilst SCAP samples have a higher mean restricted range S(obs) than the unprotected sites (8.6 to 8.3 respectively) the ranks of sites were not significantly different (Mann-Whitney U: $p > 0.05$). Overall, the SCAP was found to contain 23 of 30 restricted range species amongst the sites surveyed, corresponding to an efficiency of 77%. If the seven sites with the highest restricted range S(obs) were chosen to replace the seven SCAP sites surveyed, efficiency in representation of restricted range species would be reduced by one species to 73%. It appears that simple diversity statistics are no more likely to lead to efficient conservation of restricted range species than are *ad hoc* methods.

Figure 9.7 TWINSPAN analysis of the restricted range tree diversity of fifteen dry forests sites in Oaxaca, Southern Mexico. Sites 1 - 7 (bold) correspond to the SCAP, sites 8 - 15 are unprotected.



The TWINSPAN characterisation of forests based on restricted range species (figure 9.7) resolves five forest types after three divisions. One of these forest types, Group 1.2.1 was not found to be represented in the SCAP (80% representation, 43% redundancy).

Table 9.5. Reserve network selection for restricted range species from15 sites in Oaxacan dry forest using the Greedy selection algorithm.

	Within-SCAP selection for restricted range (7 sites only)		Sites selection to achieve 100% restricted range species representation		Sites added to SCAP to achieve 100% restricted range species representation	
Order of selection	Site	Cumulative species	Site	Cumulative species	Site	Cumulative species
1	5	15	11	17	SCAP	23
2	2	19	8	22	8	27
3	6	22	2	25	11	29
4	3	23	6	28	10	30
5	1	Not selected	5	29	9	Not selected
6	4	Not selected	10	30	12	Not selected
7	7	Not selected	1	Not selected	13	Not selected
8			3	Not selected	14	Not selected
9			4	Not selected	15	Not selected
10			7	Not selected		
11			9	Not selected		
12			12	Not selected		
13			13	Not selected		
14			14	Not selected		
15			15	Not selected		

Table 9.5 shows that consideration of restricted range species is likely to reduce the cost of conservation; only six of the fifteen sites were needed in order to ensure representation of all the restricted range species, compared to the eleven required to ensure representation of total species. Similarly for the GAP analysis, when the SCAP is taken to be non-negotiable, only three further sites were required to conserve all restricted range species surveyed, whilst an additional six were required to the SCAP for complete representation of all species. As with site selection for total species, the additional restricted range species added by the later site selections is very small.

The SCAP and Huatulco National Park

The composite checklist for Huatulco National Park comprised 134 species (dbh ≥ 5 cm). The National Park added an additional 35 species to the total list of species surveyed in the SCAP. The SCAP can therefore be said to be 74% efficient in supporting the conservation of dry forest diversity in the Park. In addition to the known species in the Park the SCAP added 29 species to the list of those species that can be considered protected in and around the municipality. The Greedy algorithm selected six of the seven SCAP sites to ensure representation of each of those additional species in the reserve network (Table 9.6). This corresponds to a redundancy of just 14% in the SCAP implying that the majority of its sites make an important contribution to dry forest conservation in Oaxaca. The assessment of the SCAP changes in relation to different objectives. If its role is seen as supplementing the National Park by provision of alternate habitats for species found in the park it is 74% efficient, if its role is to complement the park, then six out of seven of its sites add new species to the composite park list and its efficiency can be said to be 86% (100 – 14%). Either way, the *ad hoc* SCAP makes a considerable contribution to conservation of tree species diversity within the area.

Table 9.6. Potential of community managed dry forest reserves in Oaxaca to contribute to species conserved by the Huatulco National Park.

	HNP and SCAP	
Order of selection of SCAP sites	Site	Cumulative species
	HNP	134
1	6	147
2	5	153
3	7	157
4	1	160
5	2	162
6	4	164
7	3	Not selected

9.4 Discussion

The expansion of Mexico's reserve network in general, and in Oaxacan dry forest areas in particular, is of more than just of theoretical interest. In a recent survey the Mexican Commission for the Understanding and Use of Biodiversity (CONABIO) proposed the establishment of 151 new reserves for Mexico including expansion of protection for dry forest areas in Coastal Oaxaca (Cantú *et al.* 2004). The results presented here indicate that the existing *ad hoc* dry reserves of southern Oaxaca already make an important contribution to dry forest biodiversity conservation within the region and should be included in future national surveys of this type.

Whilst there is broad agreement between the orders of priorities suggested by each of the simple diversity scores, they do not appear to offer much improvement over the existing *ad hoc* site selection in terms of coverage of the forest types identified by TWINSPAN. This is perhaps not surprising given that TWINSPAN classifies samples according to the identities of species rather than simply by number of species and their abundances. The three scores therefore bias against the lower diversity sites, such as sites 1 and 6, despite their distinctive species assemblages. The more biologically random nature of *ad hoc* selection removes this bias. Howard *et al.* (1998) demonstrate the importance of this in Ugandan reserve selection where the addition of less species rich sites increased the complementarity of the overall reserve design.

Theoretical advances in reserve selection through algorithms that select sites based on complementarity have largely superseded these unsophisticated scoring methods. Here the results highlighted a similarity between the groups created by TWINSPAN and the selections of the Greedy algorithm, the latter tending to select alternately from different groups. The efficiency of the *ad hoc* reserve selection in representing the list of known species was therefore predictable from the degree of evenness in the dispersal of SCAP sites amongst the TWINSPAN defined groups. It is important to note that there was no redundancy within the SCAP as all sites contributed some species. However, use of the Greedy algorithm suggests (table 9.2) that for the 'cost' of seven reserves, more species could be represented if the SCAP were to be reconfigured, or that alternatively, additional unprotected sites would be needed if all known species from the municipality were to be added to included in reserves. This demonstrates the 'biological' inefficiency of *ad hoc* selection, and that whilst the addition of sites will increase representativity, no matter how they are selected the reserve network will remain inefficient compared to one designed from the outset on the basis of complementarity (Margules & Nicholls 1987; Pressey *et al.* 1996). That is, systematic reserve selection would result in the maximum number of species for a fixed number of sites.

In contrast, the SCAP can be considered a practical solution to the forest conservation needs of local residents, a reasonable assumption since they are responsible for its existence. For these residents, maximising species representation in the reserve network is a less important consideration than the value of soil and water conservation, continued access to forest products and the opportunity cost of the protected land. Any changes to this network might result in a loss of efficiency from the local perspective. To increase representativity of known species from 86% to 100% requires a near doubling of the size of the SCAP from 7 to 13 sites. This would imply a considerable extra investment and may not represent value for money. The long-term maintenance of tree diversity in the municipality might be better served by improving the management of the current reserves or even by investing in the conservation of other habitat types. This is, in effect, a restating of the principle that Janzen proposes of paying 5% of biodiversity to save 95% (1992 p. 31). In short, the biologically defined efficiency of the selection revealed by the selection algorithm does not necessarily correspond to, and may even run counter to, a more broadly defined cost effectiveness that takes in to account the socio-economic reality in which this conservation initiative took shape.

By more precisely defining the conservation goal as the representation of all species considered threatened, in this case all restricted range species, the goal of 100% representation was shown to become much easier. The SCAP itself was found to have some redundancy with three sites adding no species to the list. The reserve network chosen without the restriction of the SCAP required just six sites whilst by taking the SCAP as given, i.e. non-negotiable, just three more unprotected sites were required to represent all the restricted range species. Thus by concentrating on threatened species alone, biological criteria for reserve selection might more readily be incorporated into *ad hoc* reserve networks.

For both the analysis of restricted range species and of all species, a reserve selection based on one reserve from each of the groups selected by TWINSPAN was shown to approximate the selections suggested by the algorithm. Whilst such a selection would not guarantee 100% representation of species, it does have the advantage of providing nearly equivalent forests, from within the same TWINSPAN groupings, useful in the negotiating processes that accompany the 'real world' selection of reserves. The precise combination of reserves chosen by an algorithm might not always prove to be a practical option, but substitution of sites from within a TWINSPAN group might provide viable alternatives with only modest reductions in the number of species included in the reserve network.

Perhaps the issue of most concern raised by these results is the decline in mean abundance of species added to those in a reserve network as selection proceeds. Accurate estimates of abundance are increasingly difficult to obtain as species become more locally rare in a landscape and in extreme cases locally rare species will be missed in sampling exercises. The implication of this is that major land use changes might be proposed based on inadequate information as, later in the selection process, whole sites will be added to the reserve network in order to incorporate one or few locally rare species. This is likely to be true of any selection algorithm that works on the basis of complementarity, the commonest species will always be amongst the first species added to the list simply because they are likely to be found in a greater proportion of the sites under consideration for selection. That the locally most abundant species also have the widest distributions is considered to be a common phenomenon (Brown 1984; Gaston 2003) and so is likely to effect complementarity based reserve selection elsewhere. Furthermore, species with narrower distributions (endemics) are often precisely the species of concern for conservation initiatives.

TWINSPAN may be superior in this respect as it discriminates between samples based on suites of species, and may therefore be less susceptible to a very few locally rare species. In this context the algorithm may not indicate an optimal reserve network but instead indicate species that require a greater sampling effort. Those species that are apparently absent from the SCAP but present in unprotected sites may in fact have been in the SCAP but were too uncommon to have been encountered in the assessment. Verifying the presence or absence of these species in the conserved areas may be a far more cost-effective way of reaching a higher level of representation than the inclusion of whole new sites in the network. More intensive rapid assessment may be an answer but as conservation decision making will also be based on imperfect data (Pressey & Cowling 2001) such problems will always present themselves as ever less abundant species are captured by ever more intensive sampling until a highly costly total species inventory has been achieved.

A little-discussed problem in the literature on reserve selection is that of species occurring outside of their preferred habitat. Here for example, the most efficient reserve design to include all species required 11 sites (Table 9.3). Amongst the species that are added later in the selection process to the list of species accumulated are some that are more commonly associated in this area with semi-deciduous forests (*Hymenea courbaril, Guarea excelsa*) or savannas and farmland (*Enterolobium cyclocarpum*). These species are therefore amongst the drivers of the selection process but perhaps should not be priorities for dry forest conservation as their effective conservation is likely to be best advanced by consideration of other habitats. The unthinking application of systematic reserve selection is therefore not a substitute for an understanding of the diversity of an area.

Systematic reserve selection is based on a false dichotomy of species being conserved in a reserve and not conserved outside of them. Many species thrive, and will undoubtedly continue to thrive, in agricultural landscape outside of reserves, both in the particular case of Oaxacan dry forest (Gordon *et al.* 2004), and more generally (Vandermeer & Perfecto 1997). Should such early successional species as *Cordia alliodora, Cochlospermum vitifolium* and *Guazuma ulmifolia* be included in selection exercises carried out here? These species are locally common and not considered global priorities and therefore might justifiably be eliminated from consideration- as they were in the analyses limited to restricted range species. However, among the restricted range species are species such as *Cordia eleaganoides* that are locally common and, in this case, positively selected for by farmers because of its high timber value. On these grounds, *Cordia eleaganoides* could also be removed from the selection process. Again a more intimate understanding of local diversity, including knowledge of local resource management practices, rather than over-reliance on 'black-box' techniques such as selection algorithms, is required if efficient conservation is to be achieved.

There is an emerging consensus that systematic reserve selection must be seen as an indicative rather than a prescriptive aid to reserve selection (Pressey 1994). However, here it has been argued that there may be more conservation value in *ad hoc* or opportunistic reserve networks than has previously been acknowledged. In the case study presented, not only did the *ad hoc* reserve contain a significant proportion of the diversity of interest, but it also reflected local concerns and priorities and hence has local support. In proposing alterations to this or similar reserve networks, we would have to be convinced that greater species representation would not be offset by social disruption that might make the reserve network unpopular and ultimately unsustainable. Given this, it is suggested that the terms *systematic* and *optimal* may require reconsideration when used to describe theoretical reserve selection techniques that are compared to *ad hoc* methods. The negative connotation of *ad hoc,* as politically expedient and sub-optimal, may be unjustified in its implication that scientific reserve selection is necessarily better than a more 'public' one (Hull & Robertson 2000; Robertson & Hull 2001). This does not imply that systematic reserve selection has no useful role, its concentration on the fundamental units of biodiversity, whether species or habitats, is welcome where this is entirely absent, but it should not be allowed to override all other concerns and interests related to the conservation of nature.

Chapter 10: Conclusions

10.1 Introduction

This thesis has been about the translation and manipulation of an idea, or concept, called biodiversity. It has tried to show how it is manifested in different ways and how the ways in which it is accounted for are related to the networks through which it travels. In so doing it has attempted to reveal some of the reasons why the practice of biodiversity conservation has failed to meet the expectations of many biodiversity scientists. In particular, it has considered the lack of use of local scale biodiversity assessment that would otherwise be required to inform 'systematic' reserve site selection (Pressey *et al.* 1993). Here, previous conclusions are reviewed and some ways forward are suggested through which the concept of biodiversity might be more effectively put to use in a world where nature conservation and human welfare are inextricably linked.

In confronting the biodiversity crisis it is essential to understand the potential for conflict between global and local values. Much of the benefit of conserving biodiversity is expected to accrue globally, hence the rise of biodiversity conservation institutions with global reach, beyond the localities where biodiversity conservation is enacted. At these localities where critical biodiversity has been identified, the opportunity cost of conservation may be great whilst its loss may represent no more than an uncosted externality. Consequently, the needs of local and global can come into conflict.

10.2 Review

Song and M'Gonigle (2001) are correct in stating that scientists have provided the language and techniques for the physical management of the world and yet the role of science in the biodiversity crisis has been far more than that of the disinterested provider. The role of biodiversity scientists has been profound. They have been influential in framing the terms of reference by which the conservation of living nature is now discussed, practiced and enacted, and have increasingly taken on the role of advocates in addition to their role as researchers (e.g. Meffe 1999). Tacaks (1996 p. 7) describes the authority with which they are now invested:

> Anyone interested in the dwindling resources *biodiversity* represents must turn to conservation biologists for guidance. In the name of biodiversity, biologists hope to increase their say in policy decisions, to accrue resources for research, gain a pivotal position in shaping our view of nature, and, ultimately stem the rampant destruction of the natural world.

Despite this, many members of that same epistemic community continue to lament society's inability to make conservation work as they think it should (e.g. Brandon *et al.* 1998; Pressey 1994; Terborgh 2000). This thesis has looked at some aspects of this paradox.

Chapter 4 considered the practice of biodiversity assessment, its ontological foundations and how it forces a juxtaposition of local and global understandings of the species. It was argued that the supposed advantages of scientific taxonomies over local ones may in fact be less marked than is commonly supposed; both may have considerable ability to cross cultural boundaries. The advantage of scientific taxonomies lies less in any inherent properties and more in their ability to enter into scientifically informed political discourses. However the cost of this is that those discourses are inaccessible to the people most likely to be affected by the enactment of conservation - the inhabitants of biological hotspots such as Oaxaca.

One of the principal reasons why biodiversity scientists can speak with authority about biodiversity is that they can claim to *know* biodiversity better than other actors in conservation. The ways in which they come to know biodiversity were given consideration here in the examination of the epistemology of the practice of biodiversity assessment. The lack of local scale assessment to determine conservation planning is a very real manifestation of the lack of systematic conservation planning; and so it provides a useful perspective on scientific practice in conservation. Science has not arrived at a single best way of measuring biodiversity and it was shown in chapter 5 that one common assessment technique has been misapplied by its own scientific standards. This, when added to other well documented problems with biodiversity assessment, (Jepson & Canney 2001; Prendergast *et al.* 1993; Sarkar 2002) reminds us that science does not have an unchallengeable hegemony over how nature should be evaluated.

Epistemological and ontological reservations do not necessarily undermine the political authority of biodiversity assessment. Following Foucault's proposition (1980 p. 133):

> 'Truth' is to be understood as a system of ordered procedures for the production, regulation, distribution, circulation and operation of statements,

Latour (1987) contends that to understand the strength of science it is not enough to understand and place faith in its epistemologies alone, but also to understand the processes by which science is socialised, how it mobilises rhetorical and physical resources to make its case. Ultimately scientific truth emerges because people are convinced by these processes. Thus we can understand why biodiversity assessment has come to be such a powerful tool in reconstructing the cartography of entire continents (e.g. Dinerstein *et al.* 1995; Mittermeier *et al.* 1998) and in directing investment in biodiversity conservation, despite the ontological and epistemological problems associated with its practice. It follows that the concept of biodiversity shares many of the properties of a 'quasi-object'; real in that it is more than just a representation constructed through social processes, but that at the same time is inseparably engaged with the social in that it is 'disputed, uncertain, collective, variegated [and] divisive' (Latour 2000 p. 119).

Biodiversity scientists, through their reconfiguration of nature as biodiversity are embarked on a project to alter 'the geographical configurations of nature' (Takacs 1996 p. 7), and this they can do because of the relationship between knowledge and power. A second proposition of Foucault (1980 p. 133) is that

> 'Truth' is linked in a circular relation with systems of power which produce and sustain it, and to effects of power which it induces and which extend it.

This argument is manifested in the practice of conservation through the use of the concept of biodiversity as Freyfogle (2001 p. 870) notes:

> The fundamental reality is that a goal phrased in scientific terms is likely to augment the roles of science and scientists in efforts to promote it. Whether or not intended, such phrasing entails a move to gain power.

The issue being dealt with here becomes one of the relation of knowledge to power, a relationship which has been explored in post-colonial critiques. Bhaba (1994) describes postcolonial thought as being about understanding and challenging the binary of centre/periphery that dominates the world and about the uneven forces of cultural representation involved in the contest for political and social authority. Biodiversity conservation, like many of the discourses subject to postcolonial critique, is part of a diffusion of institutions and technologies from the core to the periphery (Sluyter 1999). Leach and Scoones (2003) argue that internationalised concepts, in which biodiversity can be included, can have a powerful influence over local debates (even if that influence is diffused through complicated and relations between institutions and politics) to the point that local discourses can be silenced, with the result that local actors fail to get their views 'on the agenda'. Biodiversity and the biodiversity agenda appear to be examples of this, as was shown in chapter 4.

Chapter 7 explored the network of institutional relationships through which this biodiversity conservation is promoted in Oaxaca and showed that local NGOs active in conservation occupied a rhetorical space between rural populations and the biodiversity agenda. The same chapter also confirmed the contention of Brand and Görg (2003) that to understand biodiversity we also have to understand the institutions though which it travels. In the context of biodiversity prospecting (a commercially orientated form of biodiversity assessment) they argue that (p. 231):

> It is possible to understand the regulation of biodiversity as a process where different actors and their interests confront each other, building through their compromises relatively stable institutions and management forms [...] How the interests of weaker actors and protection concerns are integrated in the institutional forms is a question of strategies, coalitions and power relations.

Their concern, like that of Leach and Scoones (2003), is for the weaker actors in the politics and practice of biodiversity conservation. Chapters 7 and 8 considered the role of local NGOs in Oaxacan conservation and concluded that, despite the epistemic and financial authority of their funding partners, local NGOs could not easily be allotted the role of the weaker partner. The space they had come to occupy in the network was a position of great strength as they could claim access to a very important resource, that of knowledge of 'the local'. Given their financial backers' allegiance to the rhetoric of partnership, they have been allowed to become gatekeepers to the biodiversity hotspots of Oaxaca. To reinforce their credentials as keepers of knowledge, they partook of 'practices of justification' including the maintenance of GIS capabilities and the accumulation of botanical specimens in herbaria. In this space, which they had largely created for themselves, they have reconfigured biodiversity, perhaps with the tacit collusion of their funding partners, as a landscape phenomenon better suited to their agendas for agrarian development.

The result of this, it was argued, was a network of negotiation, in which biodiversity conservation had been extended into Oaxaca but had not achieved a hegemonic status amongst Oaxaca's conservation NGOs. Latour (1987) suggests that the successful application of science is characterised by the progressive extension of a network whilst failure is characterised by a punctured network. Here the obvious conclusion is that science has failed as Oaxaca's NGOs have gone about their activities without following scientific prescriptions, reconfiguring biodiversity to their liking whilst using the representational practices of herbaria and GIS to give the impression of scientific and technological sophistication.

Should this reconfiguration be taken to signify a complete failure of the biodiversity agenda? First it can be noted that biodiversity is acknowledged even by some within the conservation biology community as being one of several concepts vague enough that they can be defined as they are used (Freyfogle & Lutz Newton 2001), it is perhaps more realistic to speak of *several* related biodiversity agendas. It is suggested here that a sharp distinction between the success and failure of the conservation sciences in Oaxaca is too simplistic. Biodiversity scientists have had considerable influence on Oaxacan conservation. International interest in this state is a direct result of their regional level assessments and the rhetorical power of the biodiversity agenda. Furthermore as was shown in chapter 9, the *ad hoc* approach to site selection in at least one of Oaxaca's priority ecosystems has led to a considerable proportion of the most important tree species falling into protected areas. If it is a failure, it is only partially so, and perhaps only in the eyes of the biodiversity scientists themselves.

10.3 Political ecology, Post-colonialism and biodiversity conservation in Oaxaca.

10.3.1 Political ecology

A field of critical enquiry that has more directly addressed itself to the management of natural resources, and particularly with respect to rural livelihoods in the less developed South, is political ecology. Wilshusen (2003 p. 42) suggests that political ecology:

> [D]oes not present a coherent framework or set of theoretical propositions. Rather it presents related areas of intellectual enquiry drawing on both perspectives from both the natural and social sciences.

However, the characteristics of political ecology can be identified. Borrowing from political economy, political ecology links political and economic factors in explaining patterns of resource use and environmental change (Stedman-Edwards 2000). As such it attempts to provide socio-economic explanations of phenomena that political economists contend could not be adequately explained in techno-scientific terms. It brings politics into nature. In redressing this imbalance political ecology has been criticised for tending to uncritically assume that influences from the wider political-economic system are necessarily the determinants, and often malign determinants, of local resource use whilst neglecting more local and ecological explanations. Indeed the lack of ecology in political ecology is notable (Vayda & Walters 1999). Against this critique of too little ecology and too many monolithic macro-political determinants, Latour (2004) questions the belief that politics has ever left nature in the first place. The investigation of Oaxacan biodiversity conservation presented here is, in many respects, a political economic critique. It is centred on a consideration of biodiversity as a discursive practice capable of structuring understandings of, and interaction with nature. Chapters 4 and 8 concur with Escobar (1999) in showing that there are competing ways in which biodiversity can be constructed. An inevitable consequence of this is that biodiversity has the potential to be a source of conflict when one construction contravenes the understandings of other actors with a claim to an interest in nature (Wilshusen 2003).

However the critique offered here argues that a particular macro-political influence, the *biodiversity agenda*, has not achieved hegemonic influence in Oaxaca. The interaction between Oaxaca's local NGOs and this agenda was shown to be more complex, with biodiversity being the contested concept at the centre of their relationship with their national and international funding partners. Biodiversity did not create an arena for conflict, but rather a large enough rhetorical space for the fashioning of a reconfiguration of biodiversity which in practice has proven acceptable for both parties, if not to distant biodiversity scientists. In the process, a direct translation of the concept of biodiversity from academia to Oaxaca's forests has become impossible, with the consequence that local scale biodiversity assessment has become practically irrelevant (but see below) in their pursuit of landscape conservation and agrarian development.

The planning of nature conservation (the assessment and selection of reserves) was never likely to be achieved by scientific prescription alone, especially with such a narrow reading of nature as that of biodiversity. For this to have been possible would have required conservation landscapes devoid of all cultural reference points on which the biodiversity agenda could be played out. This is simply an unrealistic expectation that has proved illusory in North and is less likely still in the South. For better or for worse the prescriptions of scientists are now contested as perhaps never before, and as the controversies over the genetic modification of organisms and biodiversity prospecting demonstrate, this is as true in the South as in the North (e.g. Nigh 2002; Scott 2003).

10.3.2 The Postcolonial critique

The partial failure of the translation of biodiversity, viewed through the lens of postcolonial criticism, may be welcome as it implies the overturn of the imposed hierarchies where the transfer of technology has not resulted in:

> The transfer of power or the displacement of a neo-colonial tradition of political control through philanthropy' (Bhabha 1994 p. 247).

The criticisms provided by the postcolonial reading of environmental regulation, of which modern biodiversity conservation is a part, (Escobar 1999; Guha 1989;2000; Vidal 2001) are not without foundation. Indeed the Northern construction of the 'tropics' as a biodiverse place in which conservation ought to be enacted has much in common with previous colonisation processes which constructed the tropics as a place of abundance on which other forms of socio-economic ordering and development should be imposed (Naylor 2000; Sluyter 1999;2001). Against this

tendency, the efforts of Oaxaca's NGOs to reconfigure biodiversity can be seen as an attempt to make a place for humans in biodiversity conservation. Similar issues have been worked through and fought over in places that have more lengthy and complicated histories of institutionalised conservation than has Oaxaca, and lessons could be learnt from the UK (Adams 1996) where biodiversity conservation is only a part, albeit a very important part, of the politics of managing nature. There is however a tendency on the part of biodiversity scientists to forget these lessons, as if the tropics are fundamentally different from elsewhere, and yet internally uniform (Driver & Yeoh 2000). The call for more systematic planning in tropical conservation, where systematic implies much greater reliance on biological criteria is an example of this.
Given the cultural distance between the Northern based conservation institutions and biodiversity hotspots of the South; it is inevitable and necessary that authority of biodiversity conservation is questioned and its discourses at least modified and adapted on arrival in the South. This does not equate to saying that the biodiversity agenda has no legitimacy in rural Oaxaca, but instead that it has to earn legitimacy anew in peripheral places such as this. The scientifically constructed authority of the biodiversity agenda has to be supported by a process of the social construction of authority (Brechin *et al.* 2003) and given that the enactment of biodiversity conservation is always local, this must be 'embedded within a larger conceptualization of the space around the local place' (Norton & Hannon 1997 p. 234).

10.4 Recommendations for biodiversity assessment in Oaxaca

Given that biodiversity assessment has succeed in claiming the authority that scientific practices are traditionally accorded, the question now is how can it be made relevant and accessible in Oaxacan conservation.

Part of the solution lies in redefining the role of biodiversity assessment, and by extension the role of biodiversity conservation. The compositionalist view of nature (Callicott *et al.* 1999) as biodiversity helps to define a disciplinary territory but opens a gap between it and the:

> [S]ocial construction of a world which allows the products of the discipline to make history with social, economic political and industrial interests. (Stengers 2000 p. 188.9)

Stengers (ibid) goes on to argue that making this history has proved problematic for all the scientific disciplines. Biodiversity scientists therefore have to re-engage with the socio-economic interests that define hotspots just as much as do the biological criteria. Instead of positioning biodiversity conservation as *the* solution to what they perceive of as detrimental practices, it needs to be reconceptualised as one relevant socio-economic process amongst many others. The practice of biodiversity conservation must foster a sense of place as shared and communal rather than a sense of detachment that results when biodiversity assessment is used to biologically imagine communities (Bryant 2002). This does not deny a role for the biodiversity agenda in Oaxaca, far from it, biodiversity loss is a real and pressing concern. However the legitimacy of biodiversity conservation in a given place should be measured by the degree to which it is engaged with local needs and concerns, not by the degree to which scientific prescriptions are enacted uncritically. Negotiation therefore has to replace prescription.

In some cases what might previously have appeared to have been failed conservation, given the degree to which local concerns appear to have outweighed global concerns, may instead represent a success in allowing more local interpretations to have greater sway over land use allocations. Satisfying local needs is ultimately more likely to determine the long term sustainability of a conservation intervention. In short, biodiversity scientists must accept a more modest, more negotiated set of criteria against which to measure the progress of biodiversity conservation.

This is an argument for the more (local) people orientated conservation that has recently come under attack from some biodiversity scientists for not delivering the biodiversity protection they have expected (Alpert 1996; Terborgh 1999; van Schaik & Rijksen 2002). Defenders of what is variously known as integrated conservation and development projects (ICDPs) and community based conservation (CBC) point out that these criticisms overlook important aspects of the social and political processes in which conservation is embedded (Brechin *et al.* 2002). The point is, however, that just as overly prescriptive applications of conservation science are unlikely to be considered legitimate by local actors because of their relation to foreign concepts and their exclusion of localised discourses, so too must room be found for the global concerns, which scientists attempt to speak for, to be heard in local development discourses.

Chapter 9, provided an example of this, showing firstly that an *ad hoc* reserve system did indeed contain a good percentage of the species found in the area. Rather than suggest that this system (which came into existence because local communities wanted it) should be redesigned, a more appropriate use of biodiversity assessment would be to direct improvements to it through negotiation with the same communities.

When global concerns are confronted at local scales they need to be presented as adaptive rather than hegemonic. This implies bottom-up and top-down flows of influence in which the local NGO's of Oaxaca should have a crucial role as sites for negotiation. This thesis has shown them capable of contesting concepts received from global institutions. The degree to which they are engaged in negotiation rather than prescription with their client groups, Oaxaca's agrarian communities, has not been the subject of this thesis but would be the obvious point of departure for further research. It is likely that different NGOs would be found to behave differently in this respect.

Above, biodiversity assessment was described as *practically* irrelevant, the qualification being important because, despite the landscape reconfiguration of biodiversity prevalent in Oaxaca, there is amongst some NGOs a desire to know more about the species they are surrounded by. Evidence of this is found in the growth of the number of herbaria in Oaxaca and the products of species level investigations by these NGOs that have begun to emerge (Gordon *et al.* 2005; Salas-Morales *et al.* 2003). However, species level assessment of biodiversity, as practiced by scientists is technically and financially demanding thus the reconfiguration of biodiversity as a more tractable landscape phenomenon, does not necessarily imply an ideological rejection of a more compositionalist approach, but a pragmatic means of moving forward.

There remain powerful arguments in favour of the wider application of local biodiversity assessment. Scientifically rigorous biodiversity assessment can aid the agrarian development agendas of Oaxaca's NGOs by precisely defining the species and places that are most critical for conservation thus simultaneously reducing the burden of conservation on local residents to an absolute minimum, and attracting the resources that biodiversity conservation institutions have at their disposal. Indeed there is likely to be a considerable advantage for local NGOs in having control over the means and results of biodiversity assessment when negotiating with national and international biodiversity conservation institutions. This does not require an abandonment of the emerging reconfiguration of biodiversity as landscape, it requires only that it too is explained and negotiated along with all other representations.

Recommendations emerging from this work with respect to the practice of biodiversity assessment relevant to local conservation NGOs are as follows.

- Biodiversity assessment is a *guide* for improving planning; it alone cannot decide optimal conservation strategies. It should therefore be seen as a tool to provide a certain kind of input into much wider processes of negotiation concerning rural land use planning.

- Biodiversity assessment *is* a scientific practice and if scientific norms are not observed an assessment exercise will have a reduced impact amongst an important groups of actors; biodiversity scientists. Thus good sampling strategies, statistical robustness and confident species identification must be prioritised.

- Knowledge is power. It is important that local NGOs maintain control over the outputs of assessment, thus whilst collaboration with scientific institutions will often be necessary and desirable, (see the previous recommendation) to ensure effective use of assessment outputs it is important that local NGO's, rather than their national and international collaborators, maintain control over their aims, methods, timing and delivery.

- The most widely used assessment techniques are not necessarily the best for local scale assessments. It is important that assessments answer questions relevant to local decision making rather than questions more relevant to academic advancement, in particular comparability with previously published studies from elsewhere may be irrelevant.

The need for interventions to be negotiated with local actors places a considerable burden on local NGOs particularly as the concept of biodiversity remains alien to those actors. Two ways are suggested in which it can be eased. The first would be to critically evaluate the degree to which representational practices such as GIS and herbarium management are really useful in promoting either conservation or agrarian development. It is suggested that they have come to be used primarily for display rather than analysis, and there is now duplication of these practices in Oaxaca that could be reduced with resources then released for biodiversity assessment. The second requires a rethinking on the part of Mexico's biodiversity scientists of their own role in grounded research and practical application of good assessment practices. It is vital that when assessment is carried out it is of sufficient scientific rigor that it can speak on behalf of Oaxacan biodiversity to the international conservation interests who, despite postcolonial critique, remain influential actors in global conservation. Constructive engagement would be an appropriate mode of negotiation. Mexico's scientists, positioned between international and local actors, can play a significant role in this if they are able to choose to prioritise supporting people orientated conservation over academic pursuits. Unfortunately, their international scientific collaborators are often demanding of work that is considered internationally relevant.

What biodiversity assessment cannot be is the single arbiter of land use planning onto which social and political processes are grafted. Instead, it has to provide one 'voice' that is both heard and understood by relevant actors negotiating land use planning. In this way it may influence but not determine the direction of on-going social and political processes. What biodiversity scientists might then achieve is an improvement on *ad hoc* planning, and one whose legitimacy is more widely accepted.

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Appendix 1. List of Forest Sites Surveyed

Site	Name	Sites in Chapter 5	Sites in Chapter 6	Sites in Chapter 9	Protection	Latitude (N)	Longitude (W)
1	Cerro Chacalmata	-	-	1	SCAP	15° 51' 01"	96° 18' 01"
2	La Jabalina	1	1	2	SCAP	15° 48' 52"	96° 06' 49"
3	La Aurora	2	2	3	SCAP	15° 47' 59"	96° 14' 58"
4	Huatulco National Park	3	3	*	National Park	15° 46' 06"	96° 11' 34"
5	Huatulco National Park	4	4	*	National Park	15° 46' 19"	96° 11' 52"
6	La Ceiba	5	5	4	SCAP	15° 49' 33"	96° 06' 58"
7	Rancho Cervantes	6	6	5	SCAP	15° 49' 09"	96° 08' 03"
8	Palo Alto	7	-	8	Unprotected	15° 49' 28"	96° 12' 23"
9	Las Palmas	8	7	9	Unprotected	15° 48' 41"	96° 10' 18"
10	El Jardin Guapinolero	-	8	6	SCAP	15° 51' 02"	96° 19' 24"
11	Rincón Viejo	-	-	10	Unprotected	15° 50' 26"	96° 07' 09"
12	La Jabalina Carretera	-	-	11	Unprotected	15° 47' 46"	96° 06' 27"
13	Rio Copalita	-	-	12	Unprotected	15° 48' 27"	96° 03' 31"
14	Faisan Viejo	-	-	13	Unprotected	15° 48' 28"	96° 11' 27"
15	Faisan Viejo	-	-	-	Unprotected	15° 48' 38"	96° 11' 22"
16	Cerro del Arenal	-	-	14	Unprotected	15° 43' 15"	96° 15' 33"
17	Cerro Desconocido	-	-	7	SCAP	15° 48' 29"	96° 20' 28"
18	Huatulco National Park	-	-	*	National Park	15° 43' 40"	96° 09' 38"
19	Huatulco National Park	-	-	*	National Park	15° 44' 01"	96° 09' 31"
20	Sector U	-	-	15	Unprotected	15° 46' 40"	96° 08' 43"
21	Sector U	-	-	-	Unprotected	15° 46' 37"	96° 18' 40"

* Data from these sites included in the National Park composite checklist.
SCAP = Communal System of Protected Areas.

Appendix 2. Checklist of Species Identified

Actinidiaceae
Saurauia scabrida Hemsl.

Anacardiaceae
Astronium graveolens Jacq.
Comocladia engleriana Loes.
Mangifera indica L.
Spondias purpurea L

Anonaceae
Sapranthus aff *violaceous* (Dunal) Saff.
Sapranthus microcarpus (Donn.Sm.) R.E.Fries

Apocynaceae
Plumeria rubra L.
Rauvolfia tetraphylla L.
Stemmadenia obovata (Hook. & Arn.) Schum.
Thevetia ovata (Cav.) A.DC

Araliaceae
Dendropanax arboreum (L.) Decne. & Planch.

Bignoniaceae
Godmania aesculifolia (HBK) Standl.
Tabebuia impetiginosa (Mart. ex DC.) Standl.
Tabebuia ochracea (Cham.) Standl.
Tabebuia rosea (Bertol.) DC.

Bombaceae
Ceiba aesculifolia (Kunth) Britton & Baker f.
Ceiba pentandra (L.) Gaertn.
Pseudobombax ellipticum (Kunth) Dugand

Boraginaceae
Cordia alliodora (Ruiz & Pav.) Oken
Cordia dentata Poir.
Cordia elaeagnoides A.DC.
Cordia seleriana Fernald

Burseraceae
Bursera excelsa (Kunth) Engl.
Bursera fagaroides (Kunth) Engl.
Bursera heteresthes Bullock
Bursera simaruba (L.) Sarg.

Cactaceae
Pereskia lychnidiflora DC.

Caesalpiniaceae
Bauhinia divaricata L.
Bauhinia subrotundifolia Cav.
Caesalpinia pulcherrima (L.) Sw.
Caesalpinia sclerocarpa Standl.
Caesalpinia velutina (Britt. & Rose) Standl.
Hymenea courbaril L.
Poeppigia procera (Spreng.) C.Presl.
Senna mollissima (Willd.) Irwin & Barneby

Capparidaceae
Capparis indica (L.) Druce
Cratevea tapia L.
Forchammeria pallida Liebm.
Morisonia americana L.

Caricaceae
Jacaratia mexicana A.DC.

Cecropiaceae
Cecropia obtusifolia Bertol.

Chrysobalanaceae
Licania arborea Seem.

Cochlospermaceae
Cochlospermun vitifolium (Willd.) Spreng.

Combretaceae
Bucida macrostachya Standl.
Bucida wigginsiana Miranda

Dilleniaceae
Curatella americana L

Ebanaceae
Diospyros salicifolia Humb. & Bonpl. ex Willd.

Erythroxylaceae
Erythroxylum areolatum L.

Euphorbiaceae
Cnidoscolus tubulosus (Muell. Arg.) I.M.Johnst.
Croton septinervius McVaugh
Sapium macrocarpum Muell. Arg.

Flacourtiaceae
Casearia arguta HBK
Casearia cf sylvestris Swartz.
Casearia commensoniana Camb.
Casearia corymbosa Kunth
Casearia tremula (Griseb.) Wright
Homalium trichostemon S F Blake
Xylosma flexuosa (Kunth) Hemsl.

Guttiferae
Calophyllum brasiliense Cambess. var. *rekoi* Standl.

Hernandiaceae
Gyrocarpus mocinnoi Espejo

Julianaceae
Amphipterygium adstringens (Schlecht.) Schiede

Lauraceae
Nectandra salicifolia (Kunth.) Nees

Malpighiaceae
Byrsonima crassifolia (L.) Kunth
Bunchosia ceroli W.R.Anderson

Melastomataceae
Conostegia xalapensis (Bonpl.) D. Don

Meliaceae
Cedrela odorata L.
Swietenia humilis Zucc.
Trichilia hirta L.

Menispermaceae
Hyperbaena mexicana Miers

Mimosaceae
Acacia cochliacantha Humb. & Bonpl.
Acacia collinsii Safford
Acacia farnesiana (L.) Willd.
Acacia hindsii Benth.
Albizia occidentalis Brandegee
Chloroleucon mangense (Jacq.) Britton & Rose var. *leucospermum* (Brandegee) Barneby & J.W.Grimes
Enterolobium cyclocarpum (Jacq.) Griseb
Lysiloma divaricatum (Jacq.) Macbride
Piptadenia obligua (Pers.) MacBride
Pithecellobium dulce (Roxb.) Benth.
Pithecellobium lanceolatum (Humb. & Bonpl. ex Willd.) Benth.
Prosopis juliflora (Sw.) DC.

Moraceae
Brosimum alicastrum Sw.
Ficus cf cotinifolia HBK
Jatropha curcas L.
Jatropha malacophylla Standl.
Jatropha sympetala Standl. & Blake
Maclura tinctoria (L.) D. Don ex Steud.

Myrtaceae
Psidium guajava L.

Nyctaginaceae
Guapira petenensis (Lundell) Lundell

Olacaceae
Ximenia americana L.

Papilionaceae
Andira inermis (Wright) Kunth
Apoplanesia paniculata Presl.
Dalbergia granadillo Pittier
Erythrina lanata Rose
Gliricidia sepium (Jacq.) Steudel
Lonchocarpus emarginatus Pittier
Machaerium biovulatum Micheli
Myrospermun frutescens Jacq.
Piscidia carthagenensis Jacq.
Pterocarpus acapulcensis Rose

Polygonaceae

Coccoloba barbadensis
Coccoloba caracascana Meesn.
Ruprechtia fusca Fernald

Rhamnaceae

Karwinskia humboldtiana (Roem et Schult.) Zucc.
Zizyphus amole (Ses. et Moc.) M. C. Johnst.

Rubiaceae

Calycophyllum candidissimum (Vahl.) DC.
Genipa americana L.
Hintonia latiflora (Sesse & Moc ex DC) Bullock
Psychotria microdon (DC) Urban
Randia laevigatoides
Randia thuberi S. Watson

Rutaceae

Esenbeckia berlandieri Baill. ex Hemsl. ssp. *litoralis* (Donn.Sm.) Kaastra
Zanthoxyllum caribaeum Lam.
Zanthoxylum fagara (L.) Sarg.

Sapindaceae

Cupania dentata DC
Sapindus saponaria L.
Thouinia serrata Radlk.
Thouinidium decandrum (Humb. & Bonpl.) Radlk.

Sapotaceae

Pouteria campechiana (Kunth) Baehni
Sideroxylon capiri (A.DC.) Pittier ssp. *tempisque* (Pittier) T.D.Penn.

Solanaceae

Cestrum nocturnum Martens

Sterculiaceae

Guazuma ulmifolia Lam.
Helicteres mexicana Kunth

Ulmaceae

Celtis iguaneae (Jacq.) Sarg.

Theophrastaceae

Jacquinia macrocarpa Cav.

Tiliaceae

Heliocarpus donnell-smithii Rose
Luehea candida (Moc. & Sessé ex DC.) M.Mart.
Trichospermum mexicanum (DC.) Baill.

Urticaceae

Urera caracasana (Jacq.) Griseb

Verbenaceae

Cythaeroxylum affine D. Donn
Vitex hemsleyi Briq.
Vitex pyramidata Robinson

Zygophyllaceae

Guaiacum coulteri A. Gray

Lippia mcvaughii Moldenke