TABLE 10.10. *Correspondence between keywords related to* metrics *and* indicators *to disciplines, activities, and land-use and land-cover related concepts*

	Metrics as percentage of total hits	Indicators as percentage of total hits
Landscape	0.5	16
Landscape change	4.3	30
Landscape architecture	0.4	32
Landscape ecology	2.9	27
Landscape planning	0.9	18
Landscape history	0.3	29
Landscape science	1.2	29
Landscape geography	0.6	22
Landscaping	0.1	19
Landscape design	0.2	19
Landscape management	0.8	21
Landscape protection	0.2	20
Landscape conservation	1.0	29
Landscape assessment	2.5	29
Landscape evaluation	3.2	31
Landscape classification	5.1	41
Land use	0.5	28
Land cover	1.2	36
Land use/land cover	3.1	33
Land-use change	1.2	19
Land-cover change	4.7	37

10.4 Discussion

10.4.1 The Internet survey

Most Internet-based analyses relate to content analysis of texts (Miller and Riechert 1994, Popping 2000, West 2001) or to technical aspects of Internet searching and tools (Litkowski 1999, Ridings and Shishigin, 2002). Internet searches of scientific databases have been used in a comparative analysis of land-use change models (Agarwal *et al.* 2002), but do not cover all domains related to landscape, such as countryside and design. The Internet survey here is more general and explorative. Hits of associated keywords do not necessarily mean a reasonable relationship among them, and the nature of the relationship remains unknown. For example, a search for *history* and *countryside* will give results including topics such as "natural history," "countryside and gardening," and "history of the countryside." The PageRank algorithm, which is similar to a citation index, only reflects the importance of the web page in terms of linking (Ridings and Shishigin 2002), and thus is not useful in defining the significance and quality of the content. However, the majority of the results obtained in the study by refined searches using combinations of keywords showed meaningful associations. The large number of hits and the important differences between the outcomes contributes to the reliability. Besides the relative importance of hits, their absolute number was also indicative for the interpretation.

Other important limitations on the use of an Internet search do exist. First, the worldwide context is severely biased by the dominant use of the English language, as is demonstrated by the occurrence of the concepts related to *landscape* (Table 10.1). Second, the Internet is a highly dynamic medium. The results refer only to one moment. However, a test made at two different times (January and May 2003) showed that an overall growth in the number of hits could be noted, but that the correspondences between the keywords used remained rather stable. The results for the different search keywords obtained at two different times were highly correlated ($r = 0.999$, $p < 0.001$), and the average growth of the number of hits was 16 percent in this period of five months. Third, it appears that Google approximates and rounds the number of results differently according to the magnitude of the search results. This makes sophisticated statistical analyses unsuitable for comparing the number of hits.

10.4.2 Dealing with land use, land cover and change

The general observation is that the term *land use* occurs more frequently than *land cover* and is used in a wide variety of contexts. Variations in meanings of these terms cannot be detected directly by this survey, but the contexts in which they are used indicate such variations. *Land use* is an important issue in activities in the landscape related to evaluation, classification, assessment, conservation, protection, and to a lesser degree management. *Land cover* appears most in relation to landscape classification, evaluation, and assessment, thus the more technical aspects on which management and conservation should be based. Land use is a more important issue in landscape ecology, landscape planning, and landscape geography, but less in landscape architecture and landscape history, which is confirmed by the very low correspondences between land use and activities of landscape design, where the term land cover is also not significant. Landscape ecology and geography use both terms most equally. On the other hand, the term land cover is relatively equal to, or in some cases more important than, land use when compared to keywords that relate to causes and processes of change. This is clear in the association with agriculture and forestry. Apparently land cover is still associated more with vegetation, although the scientific definitions also include different forms of abiotic coverage (Baulies and Szejwach 1997, Dale *et al.* 2000, Akbari *et al.* 2003). Landscape management is still poorly associated with landscape ecology, confirming the concerns raised by several landscape ecologists (Dale *et al.* 2000, Bastian 2001, Opdam *et al.* 2001).

Change is an important issue in all disciplines and activities, but remarkably less important in disciplines that intentionally create changes such as landscape architecture and landscape planning and related activities such as design and landscaping. Land-cover and land-use change are proportionally more important in association with keywords related to causes and processes, such as *nature*, *pollution*, *climate*, *fire*, and *degradation* than with other natural and environmental causes and processes. They are also highly associated with keywords such as *population* and *man*, but least of all with urbanization issues, *landscaping*, and *landscape design*, and *development*, *agriculture*, and *economy*. *Land use* is relatively more used than *land cover* in relation to *economy* and *mobility*, while *land cover* is more associated with *forestry*.

10.4.3 The context of land, landscape, and countryside

The different contexts in which land use and land cover are used indicate different approaches to the landscape. The fact that land cover is more associated with vegetation and the more technical issues of classification, evaluation, and assessment makes it more an attribute or quality of the land and less associated with broader landscape values. Zonneveld (1995) discussed the differences between land and landscape in relation to scientific disciplines. Land is more associated with soil, terrain, territory, and a series of qualities that give it a value. This value derives from an assessment of the potential land uses, and is often expressed in monetary terms. Land relates directly to land ownership. Landscape is more related to holistic and perceptive aspects of a territory, and is seen as the result of the interactions between natural processes and human activities (Council of Europe 2000). It also refers to common heritage values, both cultural and natural, belonging to a community or even to humankind, as is the case for the cultural landscape on the UNESCO World Heritage list. These values are often considered as soft values, and are rarely expressed in monetary terms.

The territory of a community is perceived through the landscape, which integrates the material qualities of the land with many nonmaterial values (Claval 2005). In rural areas this synthesis is expressed as the concept countryside, where history and culture are important aspects. To preserve these values, landscape conservation and protection, and landscape assessment are important activities. Particularly in Europe, Landscape Character Assessment (LCA) has become popular. Land use and land cover are basic data sources here (Mücher *et al.* 2003). However, the actual land use is only partially expressed by the land cover (Dale *et al.* 2000), and can be described precisely only at fine scales. Studying land-cover change refers primarily to land changes, while land-use changes embrace a broader landscape context. Modeling and predicting land-use and land-cover changes often focuses upon changes in land qualities (Fresco *et al.* 1996, Agarwal *et al.* 2002, Pontius *et al.* 2004), and less upon changes in the cultural, social, and aesthetic values of the landscape.

10.4.4 Issues not covered by the Internet survey

This Internet-based analysis allows the defining of domains of common activity, but tells nothing about the nature or scale of this activity. The associations, however, indicate different scales of action linked to the action domains of the agents involved in landscape changes. Landscape architecture and landscape planning seem to have a domain most close to the real, material actions on the ground, mostly involving individual or local agents, but with relatively little interest in the effect of the changes in a more global context and in the long term. On the other hand, sciences such as landscape ecology have a focus on land use and land cover as well as on changes over a larger spatial extent and longer duration. The link between these two domains and scales seems to be rather weak. Landscape ecologists are aware of this, and insist on better integration of ecological knowledge at all levels of action that induce or control landscape changes. Landscape management seems to be the common ground between these two domains.

Similar observations have been made in a series of studies. The Ecological Society of America, for example, proposed five ecological principles to guide decisions in land-use change and formulated eight guidelines for this (Dale *et al.* 2000). The importance of landownership in land-use change was recognized, but the focus was mainly on the USA and ecological landscape values only. A key issue is the translation of general scientific knowledge to the local agent. Agarwal *et al.* (2002) indicated that decision-making is also an important dimension as they formulated a three-dimensional framework for assessing land-use change models, including space, time, and human decision-making. The importance of scale was defined for each of these dimensions. Spatial resolution and extent in the space dimension, and time step and duration in the time dimension were proposed as well as the equivalent concepts in human decision-making "agent" and "domain." Agent refers to an individual person, a landowner, a household, a company, as well as groups of these organized as a neighborhood, municipality, region, or state. Each of these has a specific domain of action characterized by a spatial extent and duration. Six levels of complexity are recognized in human decision-making as a criterion for the assessment of land-use change models. The Land Use and Cover Change projects (LUCC) place the main problems in defining appropriate land-use change models in the lack of data, in particular, social data such as property rights and economical data on globalization (Fresco *et al.* 1996, Baulies and Szejach 1997).

Hägerstrand (1995) analyzed the connections and interactions between the micro- and macro-scale aspects of management of the biosphere, in particular how abstract knowledge can be turned into actions on the ground that cause real land-use and landscape changes. He stressed the importance of what he called territorial competence as the combination of the freedom and limitation of the landowner in making choices and considered it the most important factor at the micro scale. The choices of this primary agent depend on legal constraints, technical ability, and knowledge. Higher-order domains of human decision-making have power over larger spatial extents, but do not act directly on the landscape. They have the spatial competence to regulate and control land-use and land-use changes, but this does not necessarily correspond to reality in the landscape. He stated: "Global change is after all not the outcome of a few human actors of an immense scale. It is the nearly incalculable number of small actions which pile up to major changes in space and over time" (Hägerstrand 1995). Clearly, the contribution of sciences in this framework is not only the improvement of technological tools, but also the knowledge transfer to the domains of decision-making adapted at all scales.

10.5 Conclusions: key issues for further integration in landscape ecology

Land use and land cover are basic concepts in many disciplines and activities related to landscape research and management. They are used in many different contexts and at very different scales, which causes inevitable subtle changes in meaning. In inter- and transdisciplinary landscape studies, a precise definition of the concepts in the appropriate context is essential (Tress *et al.* 2005). Some disciplines focus on one concept. For example, landscape architecture uses mostly the term land use, whereas landscape ecology uses mostly land cover. Land use and land cover are important concepts used both as a component for characterizing landscape types and an indicator of landscape changes. Although landscape change is becoming an increasingly important issue, landscape architecture and design show a rather poor association with issues related to change and landscape dynamics. Landscape ecology is most involved with change.

Land-cover and land-use change are important in activities related to landscape protection and conservation, as well as technical issues of classification and evaluation. Land-cover and land-use changes are also important environmental processes, involving degradation, fire, pollution, climatic change, population dynamics, economical development, and particularly agriculture and forestry. Landscape change is often expressed by changes in the land cover resulting from changing natural processes and human activities, such as other choices in land use. In intensively used landscapes, the overall change of the landscape is only rarely caused by vast natural calamities; mostly it is the result of numerous small changes in discrete patches, induced by numerous agents. For many of them, not only land-cover changes but also changes in many other nonmaterial values are important. These cultural, social, economical, and aesthetical values are not only associated with the concept of landscape in its general meaning, but even more with the concept of the countryside as well.

Landscape management is the activity where landscape architecture, landscape planning, and landscape ecology seem to meet, but clearly with different perspectives. Better communication and transfer of scientific knowledge to the planners and designers seems appropriate here for more integration. Inter- and transdisciplinary studies dealing with land-use, land-cover, and landscape change can be improved by building several bridges between disciplines and activities. Looking at land-use and land-cover changes at different scales and as the integrated result of both natural processes and social, economic, and cultural needs are some of these. Transferring knowledge about these changes and the processes that induce them from scientific observations to planners, designers, and managers is another. The focus should be on linking scientific research integrated at a global scale with decision-making of agents at the local scale.

References

Agarwal, C., G.M. Green, J.M. Grove, T.P. Evans, and C.M. Schweik. 2002. *A Review and Assessment of Land-Use Change Models: Dynamics of Space, Time, and Human Choice*. General Technical Report NE-297. Newtown Square, PA: USDA, Forest Service, Northeastern Research Station.

Akbari, H., L. Shea Rose, and H. Taha. 2003. Analyzing the land cover of an urban environment using high-resolution orthophotos. *Landscape and Urban Planning* **63**, 1–14.

Antrop, M. 2001. The language of landscape ecologists and planners: a comparative content analysis of concepts used in landscape ecology. *Landscape and Urban Planning* **55**, 163–73.

Antrop, M. 2003. Continuity and change in landscapes. Pages 1–14 in U. Mander and M. Antrop (eds.). *Multifunctional Landscapes* Vol. 3: *Continuity and Change*. Southampton: WIT Press.

Bastian, O. 2001. Landscape ecology: towards a unified discipline? *Landscape Ecology* **16**, 757–66.

Baulies, X. and G. Szejach (eds.). 1997. *LUCC Data Requirements Workshop*. LUCC Report Series No.3, Barcelona: Institut Cartogràfic de Catalunya.

Brandt, J. and H. Vejre. 2004. Multifunctional landscapes: motives, concepts and perceptions. Pages 3–32 in J. Brandt and H. Vejre (eds.). *Multifunctional Landscapes: Theory, Values and History*. Vol. I. Southampton: WIT Press.

Brin, S. and L. Page. 1998. The anatomy of a large-scale hypertextual web search engine. *Computer Networks* **30**, 107–17.

Claval, P. 2005. Reading the rural landscapes. *Landscape and Urban Planning* **70**, 9–19.

Cosgrove, D. 2003. Landscape: ecology and semiosis. Pages 15–20 in H. Palang and G. Fry (eds.). *Landscape Interfaces: Cultural Heritage in Changing Landscapes*. Dordrecht: Kluwer Academic Publishers.

Council of Europe. 2000. European Landscape Convention. Firenze (http://www.coe.int/t/e/Cultural_Co-operation/Environment/Landscape/).

Dale, V.H., S. Brown, R.A. Haeuber, *et al.* 2000. Ecological principles and guidelines for managing the use of land. *Ecological Applications* **10**, 639–70.

Forman, R. and M. Godron. 1986. *Landscape Ecology*. New York: John Wiley & Sons, Inc.

Fresco, L., R. Leemans, B.L. Turner II, *et al.* (eds.). 1996. *Land Use and Cover Change (LUCC) Open Science Meeting Proceedings*. LUCC Report Series No.1., Amsterdam.

Fry, G. 2001. Multifunctional landscapes: towards transdisciplinary research. *Landscape and Urban Planning* **57**, 159–68.

Hägerstrand, T. 1995. *A Look at the Political Geography of Environmental Management*. Landscape and Life: Appropriate Scales for Sustainable Development, LASS Working Paper No.17. Dublin: University College Dublin.

Haines-Young, R. 2000. Sustainable development and sustainable landscapes: defining a new paradigm for landscape ecology. *Fennia* **178**, 7–14.

Litkowski, K.C. 1999. Towards a meaning-full comparison of lexical resources. In *Proceedings of the Association for Computational Linguistics Special Interest Group on the Lexicon, June 21–22, College Park, Maryland* (http://www.clres.com/Comparison_of_Lexical_Resources.html).

Miller, M.M. and B.P. Riechert. 1994. Identifying themes via concept mapping: a new method of content analysis. In *Communication Theory and Methodology Division of the Association for Education in Journalism and Mass Communication Annual Meeting, Atlanta, Georgia* (http://excellent.com.utk.edu/~mmmiller/pestmaps.txt).

Mücher, C.A., R.G.H. Bunce, R.H.G. Jongman, *et al.* 2003. *Identification and Characterisation of Environments and Landscapes in Europe*. Wageningen: Alterra-rapport 832.

Muir, R. 1999. *Approaches to Landscape*. London: MacMillan Press.

Nassauer, J.I. 1997. *Placing Nature: Culture and Landscape Ecology*. Washington, DC: Island Press.

Olwig, K.R. 2004. "This is not a landscape": circulating reference and land shaping. Pages 41–66 in H. Palang, H. Sooväli, M. Antrop, and S. Setten (eds.). *European Rural Landscapes: Persistence and Change in a Globalising Environment*. Dordrecht: Kluwer Academic Publishers.

Opdam, P., R. Foppen, and C. Vos. 2001. Bridging the gap between ecology and spatial planning in landscape ecology. *Landscape Ecology* **16**, 767–79.

Pontius, R.G., E. Shusas, and M. McEachern. 2004. Detecting important categorical land changes while accounting for persistence. *Agriculture, Ecosystems and Environment* **101**, 251–68.

Popping, R. 2000. *Computer-Assisted Text Analysis*. Thousand Oaks, CA: Sage.

Ridings, C. and M. Shishigin. 2002. *PageRank Uncovered* (http://www.texaswebdevelopers.com/docs/pagerank.pdf).

Steinitz, C. 2001. Landscape ecology and landscape planning: links and gaps and common dilemmas. Pages 48–50 in U. Mander, A. Printsmann, and H. Palang (eds.). *Development of European Landscapes*. Tartu: Publicationes Instituti Geographici Universitatis Tartuensis.

Tress, B., G. Tress, and G. Fry. 2003. *Interdisciplinary and Transdisciplinary Landscape Studies: Potential and Limitations*. Wageningen: Delta Series 2.

Tress, B., G. Tress, and G. Fry. 2005. Integrative studies on rural landscapes: policy expectations and research practice. *Landscape and Urban Planning* **70**, 177–91.
Veldkamp, A. and E.F. Lambin. 2001. Predicting land-use change. *Agriculture, Ecosystems and Environment* **85**, 1–6.
West, M.D. (ed.). 2001. *Applications of Computer Content Analysis*. Westport, CT: Ablex.
Zonneveld, I.S. 1995. *Land Ecology*. Amsterdam: SPB Academic Publishing.

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11

Applying landscape-ecological principles to regional conservation: the WildCountry Project in Australia

11.1 Introduction

One of the great challenges facing humanity in the twenty-first century is the conservation and restoration of biodiversity (Convention on Biodiversity 1992). In this chapter we present the landscape-ecological underpinnings of a new nongovernment organization (NGO)-driven conservation initiative in Australia, namely the WildCountry Project.

Global and national analyses highlight the extent of environmental degradation and the need for urgent protection and restoration of biodiversity (e.g., SEAC 1996, Environment Australia 2001, World Resources Institute 2001, NLWRA 2002). Such analyses also suggest that existing conservation strategies and plans are insufficient to prevent continuing losses.

The primary question, at the most general level, is: how can a conservation system be designed and implemented for Australia that is likely to maintain biodiversity for centuries to millennia? Dedicated protected areas are a core component of a nation's biodiversity conservation system. By our calculations (Fig. 11.1) only about 6 percent of Australia is in a secure protected area. There is no theoretical or empirical basis to the proposition that this level of reservation, while necessary, is sufficient for securing the conservation of Australia's biodiversity. In any case, protected area networks are largely the result of various historical contingencies rather than the principles of modern reserve design (Margules and Pressey 2000). We suggest that the percentage of Australia reserved in protected areas is unlikely to ever exceed 10–15 percent. Our calculations (Fig. 11.1) also show that about 84 percent of the Australian continent has a native vegetation cover, is outside a protected area, and is not used for agriculture or forestry. Of this 84 percent, about 56 percent

Key Topics in Landscape Ecology, ed. J. Wu and R. Hobbs.

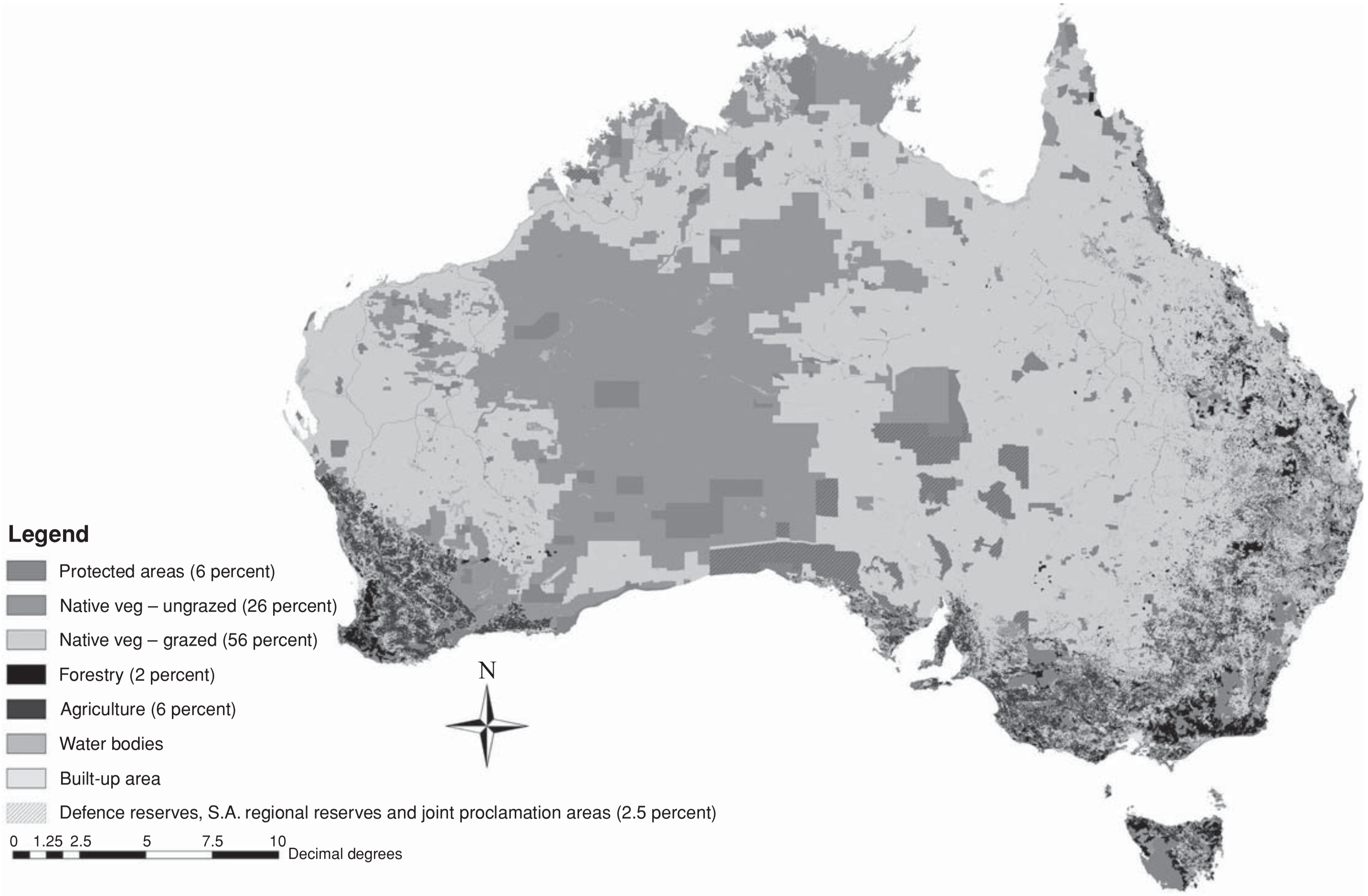

FIGURE 11.1
Broad categories of land use and land cover for Australia that identify regions where different approaches are needed for implementing landscape-wide conservation assessment and planning that promotes ecological connectivity. The legend also indicates the percentage of the Australian continent covered by each class. (Source: *1996/97 Land Use of Australia*, Version 2, National Land and Water Resources Audit.) The boundaries of the reserves that permit grazing in S.A. were extracted from the NPWSA Property Boundaries Data set from the National Parks and Wildlife S.A., Department for Environment and Heritage. Note that the protected area boundaries for NSW are current as of 2003, whereas the boundaries for Victoria and W.A. are based on the best available land tenure data which were compiled *circa* 1997.

is commercially grazed. For Australia's biodiversity to persist in the long term, more targeted and better configured reserves are needed in poorly protected country, and conservation must be integrated into the land management objectives of much of the remaining 84 percent, and especially the 56 percent of grazed, extensive country.

Civil society has now joined the government sector in attempting to formulate appropriate responses to the challenge of conserving Australia's biodiversity. As defined by international law (Convention on Biodiversity 1992), biodiversity refers to genetic, species, and ecosystem diversity, and thus encompasses, *inter alia*, the diversity found within species and the different vegetation types, food webs, and landscape ecosystems found in a region. Amidst other nongovernment initiatives such as Greening Australia (2004), The Wilderness Society Australia has launched the WildCountry Project (hereinafter WildCountry) in partnership with other civil society organizations, government at state and local levels, industry and private landowners, and the Wildlands Project USA. WildCountry builds upon the Wilderness Society's mission, namely, "to protect, promote and restore wilderness and natural processes for the wellbeing and ongoing evolution of the community of life across Australia." The WildCountry project reflects the following concepts: the need for a significant improvement in the protected area network and off-reserve management, community engagement with stakeholders to help catalyze and sustain "coalitions of the willing" capable of helping to develop and locally implement conservation assessment and planning and action on a regional basis, and recognition that assessments, plans and management must be grounded in and informed by a scientifically based understanding of what is needed to ensure the long-term conservation of biodiversity. As such, WildCountry is consistent with government policy both at the national and state level, and related conservation strategies and programs (Commonwealth of Australia 1997, Commonwealth of Australia 1999, ANZECC 2001, Commonwealth of Australia 2001a, 2001b, 2002).

The authors of this paper constitute a voluntary WildCountry Science Council, constituted in order to provide independent advice on the scientific concepts, principles, and methods needed to underpin the WildCountry project. Are existing methods for reserve design adequate? Do prevailing approaches to conservation assessment and planning provide the necessary information? Are there critical ecological phenomena and processes not yet incorporated into currently existing conservation methodologies? This paper provides an initial response to these and related questions and in so doing represents the first step in articulating a WildCountry scientific framework. In the following sections we discuss the historical and conceptual underpinnings of WildCountry and the necessary scientific principles. We conclude by considering some implications of these for WildCountry implementation.

As noted above, WildCountry assumes that, for much of Australia, voluntary changes based on partnerships between stakeholders will be the way forward. NGOs such as the Wilderness Society are well placed to help such partnerships. Governments can be constrained by inertia, vested interests or prior policy decisions. NGOs, on the other hand, can have greater flexibility and, often, greater longevity, than governments. This approach to conservation will invariably need to mesh with other programs that aim at redesigning agricultural and pastoral systems to ensure sustainability (e.g., Landcare Australia 2004). In order to facilitate such a partnership approach, education of and engagement with local communities will be key components of a WildCountry framework. Whilst acknowledging the importance of these social dimensions to WildCountry, our focus in this chapter is on the necessary scientific components of a WildCountry framework – though the social dimensions are touched upon in those sections below that address broad-scale threatening processes and approaches to systematic planning.

11.2 Foundation principles

11.2.1 Core areas

It is axiomatic that dedicated core areas must be a key component in the WildCountry framework. These are areas, primarily managed for their conservation values, that contain relatively intact ecosystems (e.g., minimal broad-scale vegetation clearing) and that have low exposure to anthropogenically driven threatening processes (however, note the discussion below on management). At a regional scale, core areas should represent all major landscapes. Another key consideration in defining dedicated core areas is the long-term prospects for retaining or improving the quality of relative wildness. Dedicated core areas must be sufficiently large to have the capacity to "self-manage" through natural processes that include the dispersal of biota and their propagules, natural selection, species evolution, and biotic regulation of local biogeochemical and water cycles (Gorshkov *et al.* 2000). There is, however, no simple answer to the question of how large an area needs be to retain core-area characteristics. Given the extent of anthropogenic perturbation in Australia (particularly in the intensive land-use areas, Fig. 11.1), we can readily anticipate that in certain landscapes it will not be possible to find large areas that have not been subject to broad-scale clearing, overgrazing, large-scale disruption of hydrological regimes, and other intensive land uses. Thus, an emphasis on linking relatively intact habitat cores that represents "the best that is left," together with substantial ecological restoration, will be necessary.

Given the importance of core protected areas to WildCountry, a logical starting point in defining the components of an appropriate scientific framework is to consider the criteria developed for the Australian Regional Forest Agreement (RFA) process (AFFA 2003). Three main criteria were adopted for the RFA, namely: comprehensiveness, adequacy, and representativeness (CAR). Comprehensiveness refers to the extent to which the pre-European distributions of forest ecosystem types are captured by the protected-area network. Representativeness refers to how well the within-forest type variability is sampled by the protected-area network. Adequacy refers to the likelihood that the protected-area network will ensure the long-term viability of the biodiversity that resides therein. In practice, the criteria of adequacy and representativeness were not substantially applied in the RFA process, and targets were only set for the first criterion – "comprehensiveness." Thus, following extensive assessments, forest tenure was changed in each region so that a nominated percentage of the pre-European distribution of forest types ecosystems was included within the protected-area network. Targets were also set to ensure a percentage of the potential habitat of threatened and rare vertebrate animal and vascular plant species were captured within the protected-area network. Interestingly, wilderness targets were also prescribed but on the basis that wilderness quality reflects a social value of no biodiversity conservation relevance.

The RFA criteria, as applied to date, have been useful in helping to promote the implementation of explicit conservation criteria and systematic reserve design in Australia (e.g., GBRMPA 2003). While they remain relevant to WildCountry, it is equally important to appreciate their limitations. The RFA criteria ignore landscape condition and thus do not explicitly consider the impact of human land-use activity on ecosystem structure and function, and animal habitat. Furthermore, landscape variation in primary productivity was not considered. Thus, in identifying priority conservation areas the distinction was not necessarily made between heavily perturbed, low productivity and relatively intact, high productivity forests.

In practice, the setting of percentage targets for representation (i.e. the comprehensiveness criterion) proved to be a relatively arbitrary process without strong and explicit scientific foundation. In any case, it is arguable whether the concept of setting percentage targets for representation is relevant in intensively cleared landscapes where only fragments of native vegetation remain. In these circumstances it could be argued that all the remnant patches have conservation value. Similarly, experience gained from studying land degradation in southern Australia has yielded little by way of guidelines as to the ecologically permissible percentage of native vegetation that can be cleared within intact landscapes. In both these contexts, the risk with a CAR approach as applied in the RFA process is to promote ecologically and numerically minimalist conservation outcomes, whereas the WildCountry conservation objectives are expansive and long-term. Nonetheless, the CAR criteria as originally conceived remain useful and relevant to the problem of systematic reserve design, and as such are one set of inputs to a WildCountry scientific framework.

11.2.2 The Wildlands Project

Additional guidance was sought from the methodology and scientific principles underlying the Wildlands Project (hereinafter Wildlands) in North America (Foreman 1999). The vision of Wildlands is to protect and restore North America's ecological integrity. The project is creating an alternative, map-based land-use plan for the continent, with the emphasis on connectivity and the restoration of ecological interactions. Formed in 1991 by scientists and conservationists, Wildlands emphasizes maintaining, connecting, and buffering wild lands, repairing landscapes that have been compromised by such factors as habitat fragmentation and loss of species, maintaining natural disturbance regimes, and communicating the ecological values of wilderness, plants, and animals (Soulé and Terborgh 1999). The approach is to restore missing species and processes, and to anticipate climatic and landscape changes that might compromise natural values and society's opportunities for enlightened economies. This is called "rewilding" (Soulé and Noss 1998). Wildlands recognizes that the application of these broad conservation principles will vary depending on regional ecology, the history of disturbance, and existing land use.

A major component of rewilding in North America is the maintenance of ecologically effective populations of large mammalian carnivores and other highly interactive species, the loss of which initiates cascading or dissipative changes through the ecosystem (Soulé *et al.* 2003). There is persuasive scientific evidence that such strongly interacting species and processes are vitally important to healthy ecosystems. Because large predators require extensive space and connectivity, the modeling of their habitat requirements is a key tool in network design in North America. Reconciling this rewilding approach with the more traditional methods of biodiversity conservation has been one of the greatest challenges for Wildlands, but is also what distinguishes its approach from that of most other conservation groups (Soulé and Noss 1998).

Following the principles of systematic conservation planning (Margules and Pressey 2000), the Wildlands regional plans feature explicit goals, quantitative targets (based on defensible ecological calculations), rigorous methods for locating new reserves, and explicit criteria for implementing conservation action. Focal species analysis can complement the incorporation of special elements and representation of vegetation types by addressing questions concerning the size and configuration of reserves and other habitats necessary to maintain species diversity and ecological resilience over time.

Wildlands provides three key concepts that are potentially relevant to the WildCountry scientific framework in Australia, namely: (1) continental and regional connectivity of large core reserves as required to support the long-term conservation requirements of large carnivores and other spatially extensive ecological processes (Soulé and Terborgh 1999), (2) complementary land management in surrounding landscapes, and, (3) where necessary, restoration of natural processes and disturbance regimes, the control of invasive species, and the reintroduction of native species. Of particular interest was the first principle, regarding the need for conservation-area designs to reflect continental and regional connectivity, the pivot points of which are large core reserves. Is this principle of large-scale connectivity equally relevant to the Australian situation, or are there major differences in the ecologies of Australia and North America that require the concept to be revisited for WildCountry?

11.2.3 Connectivity revisited

As noted above, in a North American context, large-scale connectivity has been considered by the Wildlands project in terms of the maintenance of ecologically effective populations of large mammalian carnivores and other wide-ranging focal species. The absence of large predators often leads to numerical release (abnormally high abundances) and behavioral release (e.g., abnormal levels of foraging or predation) of herbivores and mesopredators, thereby changing community composition, dynamics, and the structure of vegetation. More generally, Wildlands emphasizes the need to maintain ecologically effective populations of keystone and other highly interactive species at the regional scale (Soulé and Noss 1998, Crooks and Soulé 1999, Terborgh *et al.* 1999, Soulé *et al.* 2003).

From this perspective, planning for connectivity means ensuring large core areas to be embedded within landscapes that include compatible-use areas and habitat linkages (Frankel and Soulé 1981, Noss and Cooperrider 1994, Hobbs 2002a). It is argued that a conservation-area design based on this principle is better able to sustain the long-term ecological viability of these large species compared to a conventional system of isolated parks and reserves. This approach requires working at spatial and temporal scales exceeding those normally employed to manage natural areas and natural resources.

There are major differences in the ecologies of Australia and North America that suggest the Wildlands principle of large-scale connectivity for large mammalian carnivores may not be as relevant to WildCountry. First, and most importantly, Australia lost its megafauna around 50 000 years ago (Beck 1996). Thus, the long-term requirements of large predators might appear irrelevant in the framing of a continental conservation strategy for Australia. A second difference between Australian and North American ecology stems from the climatic systems that dominate these continents. Much of Australia is characterized by extreme variability in the distribution of rainfall as well as deeply weathered landscapes of low relief and low soil fertility. These dominating factors have generated distinctive ecological responses in the plants and animals everywhere, but particularly in the arid and semi-arid zones (Friedel *et al.* 1990, Morton *et al.* 1995).

Notwithstanding these differences, large-scale connectivity may still be an important conservation planning principle for Australia but primarily for different reasons than in North America. The following sections consider a set of ecological phenomena and processes that operate at large scales in both space and time. We argue that their ongoing functioning is necessary for the long-term resilience of landscape ecosystems, the maintenance and regeneration of habitat, and ultimately the viability of populations. Furthermore, we suggest that the landscape linkages necessary to maintain their functioning have yet to be substantially integrated into conservation assessment and planning.

11.3 Large-scale connectivity

Connectivity is generally considered in terms of wildlife corridors – narrow bands of native vegetation connecting core habitat areas (Lindemayer and Nix 1993). Here the word is used to draw attention to large-scale ecological phenomena and processes that require the maintenance of landscape linkages at regional to continental scales. The necessary landscape linkages may include core areas, comprise continuous habitat such as riparian corridors and appropriately spaced stepping-stones (Dobson *et al.* 1999, Roshier *et al.* 2001), or reflect some other kind of spatial "teleconnection."

11.3.1 Trophic relations and interactive species

Whilst Australia lacks the large mammalian carnivores of North America, species at any given trophic level can play a major role in regulating resource availability and population dynamics over species at other levels, e.g., large herbivores (Oksanen and Oksanen 2000), pollinators (honeyeaters; Paton *et al.* 2000) and mesopredators such as the dingo *Canis lupus dingo* (Caughley *et al.* 1980). Maintaining large-scale connectivity for such trophically interactive species (Soulé *et al.* 2003) is critical to consider in conservation planning.

The broader implications of maintaining and/or restoring trophic levels in a food web on a landscape-wide basis have generally not been used in Australia to guide conservation assessment and planning.

11.3.2 Hydroecology

The term hydroecology describes the role that vegetation plays in regulating surface and subsurface hydrological flows, and in turn the importance of water availability to plants and animals (Mackey *et al.* 2001). The significance in Australia of hydroecology is amplified by high year-to-year variability in rainfall (Hobbs *et al.* 1998). Hydroecological processes can be observed in all regions of Australia, including Cape York Peninsula (Horne 1995, Horne *et al.* 1995), the Southern Tablelands of NSW (Starr *et al.* 1999), the Central Highlands of Victoria (Vertessey *et al.* 1994), and inland Australia (Friedal *et al.* 1990, Stafford Smith and Morton 1990). Generally, our land management has not protected catchment-scale processes that affect groundwater recharge and discharge, although these are critical for maintaining perennial springs and water holes, river base flows, and perennial stream flow. Biodiversity conservation and planning must pay particular attention to such whole-of-catchment processes.

11.3.3 Long-distance biological movement

Both vertebrates and invertebrates can have stages in their life cycles that are associated with large-scale movement. A vast diversity of organisms and their propagules forage, disperse, and migrate (Cannon and Gardner 1999, Drake *et al.* 2001, Isard and Gage 2001). Examples of ecologically significant long-distance biotic dispersal include the use of rainforest patches by animals in Northern Australia (Palmer and Woinarski 1999, Shapcott 2000, Bach 2002), and dispersive avifauna in Australian woodlands and open-forest (Paton *et al.* 2000, NLWRA 2002). Thus, there is a need to maintain networks of suitable habitat for dispersive species over large regions. A conservation system is needed that is extensive enough to embrace the full breadth of continental variability in climate, productivity, and vegetation, and the resultant fauna dynamics (Nix 1974).

A type of biological movement of special conservation interest is dispersal to and from refugia – places where populations of a species can persist during a period of detrimental change occurring in the surrounding landscape. Thus, refugia are locations that provide refuge from threatening processes. They enable species to maintain their presence in landscapes and are potential sources for reestablishment. Refugia can be defined at a range of scales and with respect to various threatening processes, including inappropriate fire regimes (Mackey *et al.* 2002), global climate change (Lovejoy 1982), and drought (Stafford Smith and Morton 1990). Refugia are probably important in all ecosystems (thought not all movement associated with refugia is necessarily large scale) but only rarely has their significance been considered in conservation assessment and planning.

11.3.4 Ecologically appropriate fire regimes

Fire is a natural part of virtually all Australian landscapes and has an important influence on the biological productivity, composition, and landscape patterning of ecosystems (Reid *et al.* 1993, Williams *et al.* 1994, Whelan 1995, Bradstock *et al.* 2002, Catchpole 2002, Mackey *et al.* 2002). The conservation implications of ecologically inappropriate fire regimes can be substantial. In systems fragmented by human activity, broad landscape processes have been disrupted leading to altered fire regimes (Gill and Williams 1996, Hobbs 2002b). Remnant vegetation in agricultural areas may suffer from the absence of fire over long periods. In large core conservation areas, there may be an overriding need for deliberate and carefully planned fire management, allowing for large and/or high intensity wildfires. The role of Aboriginal burning practices demands special attention especially in Northern Australia (Price and Bowman 1994, Williams and Gill 1995, Bowman *et al.* 2001, Yibarbuk *et al.* 2001, Keith *et al.* 2002).

11.3.5 Climate change and variability

As a consequence of human-forced climate change (IPCC 2002), it is likely that Australian ecosystems will be exposed in the coming decades to an increase in the frequency of extreme weather events, higher average daily temperatures (especially higher minimum daily temperatures), and changes in the spatial and seasonal distribution of precipitation (CSIRO 2001). Such changes have direct and indirect impacts on all aspects of biodiversity, including species distributions, community structure, and ecosystem processes (Mackey and Sims 1993, Hannah and Lovejoy 2003, Thomas *et al.* 2004). Providing connectivity to promote biotic adaptation to climate change is a formidable challenge, but is central to continental- and regional-scaled conservation assessment and planning for the coming decades (NTK 2003).

11.3.6 Coastal zone fluxes

There are two perpendicular directions of flow in the coastal zone. One is the flux of matter and energy between sea and land; the other direction of flow is parallel to the coast, such as the migration of marine organisms, including shorebirds, and the movement of coastal currents. Connectivity of land/coastal-zone flows is particularly important given the concentration of Australia's population in coastal regions (Cosser 1997). Terrestrial conservation assessment and planning must include these important links with the marine environment. A landscape could have conservation value primarily because it contributes to ecosystem function in the adjacent coastal zone. Indeed, a "source to sea" planning framework is essential. A more comprehensive treatment of these connectivity processes will be published elsewhere.

11.4 Research and development issues

11.4.1 Dispersive fauna

Conservation planning for dispersive fauna requires data at landscape and continental scales on movements and the spatial and temporal distribution of habitat resources, including the dispersion of food resources in response to environmental variability. Meeting these information needs is conceptually tractable but logistically will require a significant investment in IT-based systems. Data from various sources (remotely sensed, field-survey records, digital maps) and themes (climate, topography, substrate, vegetation, wildlife, land use, land tenure) must be assimilated into usable formats at the best available resolutions across the continent. Advances in GIS, environmental modeling and remote sensing provide the capacity to describe, classify, and map landscapes in ways that are relevant to the assessment of fauna distributions and habitat requirements (Mackey *et al.* 1988, 1989, 2001, Lesslie 2001, Mackey and Lindenmayer 2001, Nix *et al.* 2001). They can also be used to directly track temporal variability in the distribution and availability of primary production and food resources. Critically, these analyses can now be undertaken at a continental scale with high spatial and temporal resolutions. Of particular interest are high-resolution digital elevation models (Hutchinson *et al.* 2000) and land-cover data derived from satellite-borne sensors such as MODIS (~250 m spatial resolution), Landsat TM (~25 m resolution) and JERS-1 SAR (~18 m resolution). Derived remotely sensed products now include various estimates of food resource production in response to environmental variability, including net primary productivity, above ground biomass, leaf-area index, and land-cover classes (Landsberg and Waring 1997, Austin *et al.* 2003, NASA 2003). These analytical capabilities add to existing technologies and aid in both identifying core protected areas and in designing the necessary buffers, corridors, linkages, and management changes in the surrounding landscape matrix.

11.4.2 Protected-area and off-reserve management

The design and establishment of core areas for biodiversity conservation can only be part of a WildCountry framework. Decisions must also be made about the ongoing management of such areas together with the necessary off-reserve management regimes. Management of core conservation areas will affect neighboring lands (and hence the regional community's attitude to WildCountry values and outcomes) and vice versa. In Australia, almost all lands, including protected areas, are affected by the increasing impact of feral animals and plants and altered disturbance regimes. Feral animals degrade the most remote deserts of central Australia, and feral animals and weeds transform the furthest reaches of central Arnhem Land. In the absence of preventative management, these threats drive the landscape and its natural values further into decay. It is an abrogation of responsibility to leave the conservation values of lands unprotected from the array of new elements that are altering these landscapes.

We noted above that an effective system of reserves requires high levels of connectivity either by managing the "matrix" (all areas that are not part of the network of lands and waters under some kind of biodiversity protection) to allow for the movement and dispersal of plants and animals or by creating linkages specifically for that purpose. It is unrealistic to assume that all essential connectivity can be contained within a system of reserves in isolation. It is more reasonable to assume that large areas of habitat (or landscape components that contribute to ecological function) will remain outside the reserve system. The way the matrix is managed will be critical for the long-term conservation of biodiversity (Hale and Lamb 1997, Lindenmayer and Recher 1998, Lindenmayer and Franklin 2002), including the effectiveness of the linkages needed to maintain the connectivity of large-scale ecological processes.

Off-reserve land can have a vital role to play in protecting and restoring hydrological relations, accommodating the impacts of long-term climate change, providing for the seasonal and episodic movements of animals, the dispersal of propagules, and the exchange of genetic material between core areas. For these reasons, the capacity to manage effectively will depend on the willingness of adjoining landowners and leasees to change management practices to enhance conservation outcomes. There is a growing number of examples where off-reserve conservation can serve as a key element in engaging landowners and other stakeholders in the conservation process, especially if the engagement includes the development of local capacity and understanding (e.g., Dilworth *et al.* 2000).

The challenges facing off-reserve land-use management vis-à-vis connectivity will vary depending on the environmental context, regional conservation objectives, land-use history, the degree of degradation of the habitat, and management regimes. In Australia, three broad categories of land use and land cover can be recognized (Fig. 11.1). First, there are extensive areas in the tropical north and arid, central and southern Australia that have suffered minimal clearing of native vegetation, but are now witnessing the loss of biodiversity as the result of introduced herbivores and predators, livestock, weeds, and altered fire regimes (Finlayson 1961, Morton 1990, Woinarski *et al.* 1992, Russell-Smith *et al.* 1998, Franklin 1999, Woinarski *et al.* 2001, Lewis 2002). However, this category retains the potential for effective connectivity. Second, there are landscapes dominated by agricultural production where the pre-European settlement vegetation has been largely removed, and only isolated and usually degraded remnants persist; examples include the sheep/wheat belts of southeast Australia and southwest Western Australia. The maintenance of ecological flows is far more challenging in such areas (Saunders and Hobbs 1991, Hobbs *et al.* 1993, McIntyre and Hobbs 1999). Third, there are areas that are dominated by native tree vegetation, but are subject to substantial resource extraction, in particular, forest ecosystems in southern and eastern Australia.

We cannot assume that the matrix is benign for native plants and animals. Indeed, the nature of the matrix will vary depending on the prevailing land use, and closer attention to the impact of different matrix types on species' movement and survival is needed (e.g., Davies *et al*. 2001). Some matrix areas will be ecological sinks, although species will respond differently to different kinds and degrees of disturbance, pollution, and degradation. Other areas will retain some capacity to contribute to biodiversity and the maintenance of ecosystem processes. Within the categories of landscapes just described there are significant differences in the management practices needed to restore and buffer core areas, promote ecological connectivity, protect off-reserve biodiversity, and protect on-reserve biodiversity from off-reserve hazards. Identifying the appropriate mix of complementary management practices remains an ongoing research challenge.

11.4.3 Fire regime management and social values

Management of landscapes for biodiversity conservation is not only about remedial or preventative work on invasive organisms. Effective management demands good relationships with the human communities that inhabit these landscapes. While the livelihoods of all communities in regional Australia are coupled to access to land, for Aboriginal Australians, lands cut off from people are considered "lands without life." It follows that conservation planning in the areas of Australia that are legally recognized as Aboriginal land (about 13 percent of the continent, largely but not exclusively in central and northern Australia) cannot be separated in practice from issues related to the social and economic aspirations of Aboriginal Australians. To our knowledge, Aboriginal Australians did not generally engage in broad-scale clearing or silviculture. Rather, fire was the most important component of Aboriginal land management. Substantial parts of the Australian landscape probably still reflect the impact of past Aboriginal fire management practices. In some areas, the management system persists. Understanding past and present fire regimes is a critical research task for integrating fire management into large-scale conservation planning in Australia. The challenge of integrated fire management for biodiversity conservation is no less complex when considering the management systems, values, aspirations, and rights of nonindigenous pastoralists in regional Australia.

11.4.4 Whole-of-landscape conservation planning

Significant advances have been made in identifying networks of dedicated reserves that represent some kind of optima with respect to representativeness of biodiversity at a regional scale, their spatial configuration, and the potential impact of removing land from other land uses. Systematic reserve design usually also incorporates information generated from population viability analysis undertaken for target species. The whole-of-landscape approach promoted by WildCountry suggests a similar, but more complex planning process. "Landscape viability analysis" is needed, which enables the entire landscape to be evaluated and the optimum set identified of dedicated reserves, areas of connectivity, and off-reserve management requirements.

If the problem of how to optimally allocate conservation effort can be properly formulated as a decision-theory problem then decision theory-algorithms can help solve the problem efficiently (Possingham *et al.* 2001). It is important in this context to separate the following three parts of conservation planning:

(1) *Defining the problem* in terms of the objectives and constraints – this is where the conservation values (and related socioeconomic values) that the planning is intended to promote or protect are quantified using some kind of mathematical formulation.
(2) *Describing the system state and its dynamics* – as per the target components of biodiversity and the large-scale processes discussed in this paper. This means answering such questions as: What and where are the habitat/ecosystem types? How do different activities (zoning into reserves or other uses) affect the viability of species? What are the

consequences of zoning decisions on ecological processes? And what are the consequences of spatial relationships of different human activities for ecological processes and species viability? The system state and its dynamics can include socioeconomic variables and sub-models.

(3) *Applying an algorithm* used to generate planning options. If the problem is properly defined and the system state and dynamics are adequately accounted for, then algorithms can be applied that find the best or some good solutions that aid or initiate the decision-making process. The algorithm often needs ancillary software to present alternatives and facilitate the use of potential solutions in the decision-making process. Ultimately the algorithm is no more than a decision support tool that uses computers to see possibilities that we may miss.

Traditionally the "reserve design" problem has been defined such that the objective is to minimize costs given a suite of conservation targets. However, there has been little analytical consideration of the connectivity issues discussed here. More recently, the Marxan family of software (www.ecology.uq.edu.au/marxan.htm) have been applied to solve spatial problems where the objective has a spatial component (minimize boundary length, minimum reserve size) and the targets attempt to deliver adequacy (Possingham *et al.* 2000, Possingham *et al.* 2001, Noss *et al.* 2002; also note the work of Andelman *et al.* 1999, Singleton *et al.* 2001). If issues of connectivity can be clearly defined, they can be incorporated into the algorithms. The challenge is to articulate the connectivity process issues discussed here so that they can be formulated mathematically. The existing Marxan algorithms then need to be modified to accommodate the required new kinds of objectives and constraints.

A computer-based planning tool is needed that draws upon these modified algorithms, accesses the spatial information base, and that can be used to prepare information and options for stakeholders interested in advancing biodiversity conservation in their region. As landscape viability is of equal concern for all users of the land resource, such a planning tool should be generally welcomed as a tool for meshing production and conservation objectives. Nevertheless, the difficulties with this approach should not be underestimated, as in many areas we lack basic information with which to guide landscape management, and we cannot always wait for complete information to make decisions. Simpler approaches that base decisions on partial information may stimulate activity and enthusiasm within local communities (e.g., Lambeck 1997, Dilworth *et al.* 2000). While these approaches can be criticized (e.g., Lindenmayer *et al.* 2002), they may form a useful kernel on which to build greater scientific sophistication that leads to action.

11.5 Conclusion

In summary, the WildCountry scientific framework draws from landscape ecology principles, which include the following main elements:

- Core protected-area networks must be based on systematic reserve design principles that build upon the criteria of comprehensiveness, adequacy, and representativeness, complimented by, among others, criteria related to primary productivity and landscape condition.
- Biodiversity conservation assessment and planning (including protected-area design) must move beyond traditional conservation design principles by aiming for the maintenance and restoration of large-scale (in space and time) ecological and evolutionary processes over the entire landscape. Assessments and plans must reflect the landscape linkages necessary to maintain large-scale ecological phenomena and processes related to trophic relations and interactive species, hydroecology, long-distance biological movement, refugia from threatening processes, ecological fire regimes, climate change and variability, and coastal zone fluxes.
- Proximity of the reserve system to sources of disturbance requires, as a minimum, buffering and consideration of complementary land uses and management. Whole-of-landscape conservation assessment and planning will be unavoidable; recognizing that the entire landscape (protected areas, leasehold land, Aboriginal land, unallocated crown land, private land) within which protected areas are embedded must be better managed to promote biodiversity conservation.

Regional planning must therefore include management guidelines and prescriptions for, among other things, broad-scale threatening processes including feral animals, weeds and ecologically inappropriate fire regimes, both in protected and unprotected areas. Ecological restoration in degraded landscapes will be necessary, particularly in the intensive land-use areas of Australia (Fig. 11.1). Restoration objectives should reflect the need to restore the identified large-scale connectivity processes. Landscape viability analysis will enable the entire landscape to be evaluated and the optimum set identified of dedicated reserves, areas of connectivity, and matrix (off-reserve) management requirements.

Management for biodiversity conservation that facilitates long-term ecological connectivity will remain an ongoing research and development challenge. It must be recognized that the matrix is never static, and it may be impossible to predict the quantity and quality (intensity) of development that could eventually occur on any specific parcel in any given region. Thus, the conservation utility of the matrix must be considered with caution and be recognized as complementary to dedicated core areas. In fact, it may be prudent to assume that the matrix will change, and, in the worst-case scenarios, lose all positive conservation values over time.

It must be noted that the emerging WildCountry framework described here goes beyond current reserve-based assessment and management, and hence needs a step-up in activity and funding. We acknowledge that it has proven difficult to maintain current levels of conservation management, with many agencies facing reduced budgets and having to deal with increasing threats. The challenge then is to recognize the full extent of the actions needed and convince land managers, communities, governments, and relevant agencies of the need for a broadly based landscape approach.

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References

AFFA. 2003. Regional Forest Agreements web site. Australian Government Department of Agriculture, Fisheries and Forestry (http://www.affa.gov.au/).

Andelman, S.J., I. Ball, F.W. Davis, and D.M. Stoms. 1999. *SITES V 1.0: an Analytical Toolbox for Designing Ecoregional Conservation Portfolios*. Unpublished manual prepared for the nature conservancy. (available at http://www.biogeog.ucsb.edu/projects/tnc/toolbox.html).

ANZECC. 2001. *Australian and New Zealand Environment and Conservation: Review of the National Strategy of the Conservation of Australia's Biological Diversity*. Canberra: Commonwealth of Australia.

Austin, J.M., B.G. Mackey, and K.P. van Niel. 2003. Estimating forest biomass using satellite radar: an exploratory study in a temperate Australian Eucalyptus forest. *Forest Ecology and Management* **176**, 575–83.

Bach, C.S. 2002. Phenological patterns in monsoon rainforests in the Northern Territory, Australia. *Austral Ecology* **27**, 477–89.

Beck, M.W. 1996. On discerning the case of late Pleistocene megafaunal extinctions. *Paleobiology* **22**, 91–103.

Bowman, D.M.J.S., O. Price, P.J. Whitehead, and A. Walsh. 2001. The "wilderness effect" and the decline of *Callitris intratropica* on the Arnhem Land Plateau, northern Australia. *Australian Journal of Botany* **49**, 665–72.

Bradstock, R.A., J.E. Williams, and A.M. Gill (eds.). 2002. *Flammable Australia: The Fire Regimes and Biodiversity of a Continent*. Cambridge: Cambridge University Press.

Cannon, R. and M.G. Gardner. 1999. Assessing the risk of windborne spread of foot and mouth disease in Australia. *Environment International* **25**, 713–23.

Catchpole, W. 2002. Fire properties and burn patterns in heterogeneous landscapes. Pages 49–75 in R. A. Bradstock, J.E. Williams, and A.M. Gill (eds.). *Flammable Australia: The Fire Regimes and Biodiversity of a Continent*. Cambridge: Cambridge University Press.

Caughley, G., G. Grigg, J. Caughley, and G.J.E. Hill. 1980. Does dingo predation control the densities of kangaroos and emus? *Australian Wildlife Research* **7**, 1–12.

Commonwealth of Australia. 1997. *Nationally Agreed Criteria for the Establishment of a Comprehensive, Adequate and Representative Reserve System for Forests in Australia*. Canberra: Government Printer.

Commonwealth of Australia. 1999. *National Forest Policy Statement.* (http://www.affa.gov.au/content/publications.cfm?ObjectID=CDA4CAF9-D118-4E13-AAC472AC7603EBE5).

Commonwealth of Australia. 2001a. *National Objectives and Targets for Biodiversity Conservation 2001–2005*. Canberra: Government Printer.

Commonwealth of Australia. 2001b. *Australian Native Vegetation Assessment 2001*. Canberra: National Land and Water Resources Audit.

Commonwealth of Australia. 2002. *The Regional Forest Agreements* (http://www.rfa.gov.au).

Convention on Biodiversity. 1992. Website of the Secretariat on the Convention on Biodiversity (http://www.biodiv.org).

Cosser, P. 1997. *Nutrients in Marine and Estuarine Environments*. Canberra: Australian Government Department of Environment and Heritage.

Crooks, K.R. and M.E. Soulé. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. *Nature* **400**, 563–6.

CSIRO. 2001. *Climate Change Projections for Australia*. Melbourne: CSIRO Atmospheric Research (http://www.dar.csiro.au/publications/projections2001.pdf).

Davies, K.F., B.A. Melbourne, and C.R. Margules. 2001. Effects of within- and between-patch processes on community dynamics in a fragmentation experiment. *Ecology* **82**, 1830–46.

Dilworth, R., T. Gowdie, and T. Rowley. 2000. Living landscapes: the future landscapes of the Western Australian wheatbelt? *Ecological Management and Restoration* **1**, 165–74.

Dobson, D., K. Ralls, M. Foster, *et al.* 1999. Reconnecting fragmented landscapes. Pages 129–70 in M.E. Soulé and J. Terborgh (eds.). *Continental Conservation: Scientific Foundations for Regional Conservation Networks*. Washington, DC: Island Press.

Drake, V.A., P.C. Gregg, I.T. Harman, *et al.* 2001. Characterizing insect migration systems in inland Australia with novel and traditional methodologies. Pages 207–34 in I.P. Woiwood, D.R. Reynolds, and C.D. Thomas (eds.). *Insect Movement: Mechanisms and Consequences*. Wallingford: CABI Publishing.

Environment Australia. 2001. *State of the Environment Report*. Canberra: Commonwealth of Australia (http://www.ea.gov.au/soe/).

Finlayson, H.H. 1961. On central Australian mammals. *Records of the South Australian Museum* **14**, 141–91.

Foreman, D. 1999. The Wildlands Project and the rewilding of North America. *Denver University Law Review* **77**, 535–53.

Frankel, O.H. and M.E. Soulé 1981. *Conservation and Evolution*. Cambridge: Cambridge University Press.

Franklin, D. 1999. Evidence of disarray amongst granivorous bird assemblages in the savannas of northern Australia, a region of sparse human settlement. *Biological Conservation* **90**, 53–68.

Friedel, M.H., B.D. Foran, and D.M. Stafford Smith. 1990. Where the creeks run dry or ten feet high: pastoral management in arid Australia. *Proceedings of the Ecological Society of Australia* **16**, 185–94.

GBRMPA. 2003. *Explanatory Statement, Great Barrier Reef Marine Park Zoning Plan 2003*. Australian Government, Great Barrier Reef Marine Park Authority (http://www.reefed.edu.au/rap/pdf/ES_25-11-03.pdf).

Gill, A.M. and J.E. Williams. 1996. Fire regimes and biodiversity: the effects of fragmentation of southeastern eucalypt forests by urbanization, agriculture and pine plantations. *Forest Ecology and Management* **85**, 261–78.

Gorshkov, G.V., V.V. Gorshkov, and A.M. Makarieva, 2000. *Biotic Regulation of the Environment*. Chichester: Spinger-Praxis.

Greening Australia. 2004. http://www.greeningaustralia.org.au/GA/NAT/.

Hale, P. and D. Lamb (eds.). 1997. *Conservation Outside Nature Reserves*. Brisbane: Centre for Conservation Biology, University of Queensland.

Hannah L. and T.E. Lovejoy (eds.). 2003. *Climate Change and Biodiversity: Synergistic Impacts*. Washington, DC: Conservation International.

Hobbs, R.J. 2002a. Habitat networks and biological conservation. Pages 150–70 in K.J. Gutzwiller (ed.). *Applying Landscape Ecology in Biological Conservation*. New York: Springer.

Hobbs, R.J. 2002b. Fire regimes and their effects in Australian temperate woodlands. Pages 305–26 in R. Bradstock, J.E. Williams, and A.M. Gill (eds.). *Flammable Australia: Fire Regimes and the Biodiversity of a Continent*. Cambridge: Cambridge University Press.

Hobbs, J.E., J.A. Lindesay, and H.A. Bridgman. 1998. *Climates of the Southern Continents: Past, Present and Future*. Chichester: John Wiley & Sons Ltd.

Hobbs, R.J., D.A. Saunders, and A.R. Main. 1993. Conservation management in fragmented systems. Pages 279–96 in R.J. Hobbs and D.A. Saunders (eds.). *Reintegrating Fragmented Landscapes: Towards Sustainable Production and Nature Conservation*. New York: Springer.

Horn, A.M. 1995. *Surface Water Resources of Cape York Peninsula*. Canberra: CYPLUS – Queensland and Commonwealth Governments.

Horn, A.M., E.A. Derrington, G.C. Herbert, R.W. Lait, and J.R. Miller. 1995. *Groundwater Resources of Cape York Peninsula*. Canberra: CYPLUS – Queensland and Commonwealth Governments.

Hutchinson, M.F., J.A. Stein, and J.L. Stein. 2000. *Upgrade of the 9 Second Australian Digital Elevation Model*. CRES, The Australian National University (http://cres.anu.edu.au/dem).

IPCC. 2002. *Climate Change 2001: The Scientific Basis*. Intergovernmental Panel on Climate Change (http://www.ipcc.ch/).

Isard, S.A. and S.H. Gage. 2001. *Flow of Life in the Atmosphere: an Airscape Approach to Understanding Invasive Organisms*. East Lansing: Michigan State University Press.

Keith, D., J.E. Williams, and J. Woinarski. 2002. Fire management and biodiversity conservation – key approaches and principles. Pages 401–428 in R. A. Bradstock, J.E. Williams, and A.M. Gill (eds.). *Flammable Australia: The Fire Regimes and Biodiversity of a Continent*. Cambridge: Cambridge University Press.

Lambeck, R.J. 1997. Focal species: a multi-species umbrella for nature conservation. *Conservation Biology* **11**, 849–56.

Landcare Australia. 2004. http://www.landcareaustralia.com.au/.

Landsberg, J.J. and R.H. Waring. 1997. A generalized model of forest productivity using simplified concepts of radiation-use efficiency, carbon balance and partitioning. *Forest Ecology and Management* **95**, 209–28.

Lesslie, R.G. 2001. Landscape classification and strategic assessment for conservation: an analysis of native cover loss in far southeast Australia. *Biodiversity and Conservation* **10**, 427–42.

Lewis, D. 2002. *Slower Than the Eye Can See: Environmental Change in Northern Australia's Cattle Lands: A Case Study from the Victoria River District, Northern Territory*. Darwin: Tropical Savannas Cooperative Research Centre.

Lindenmayer, D.B. and J.F. Franklin. 2002. *Conserving Forest Biodiversity: A Comprehensive Multiscaled Approach*. Washington DC: Island Press.

Lindenmayer, D.B. and H.A. Nix. 1993. Ecological principles for the design of wildlife corridors. *Conservation Biology* **7**, 627–30.

Lindenmayer, D.B. and H.F. Recher. 1998. Aspects of ecologically sustainable forestry in temperate eucalypt forests – beyond an expanded reserve system. *Pacific Conservation Biology* **4**, 4–10.

Lovejoy, T.E. 1982. Designing refugia for tomorrow. Pages 673–80 in G.T. Prance (ed.). *Biological Diversification in the Tropics*. New York: Columbia University Press.

Mackey, B.G. and D.B. Lindenmayer. 2001. Towards a hierarchical framework for modeling the spatial distribution of animals. *Journal of Biogeography* **28**, 1147–66.

Mackey, B.G., D.B. Lindenmayer, M. Gill, M. McCarthy, and J. Lindesay. 2002. *Wildlife, Fire and Future Climate: A Forest Ecosystem Analysis*. Melbourne: CSIRO Publishing.

Mackey, B.G., H.A. Nix, and P. Hitchcock. 2001. *The Natural Heritage Significance of Cape York Peninsula*. Canberra: ANU Tech P/L.

Mackey, B.G., H.A. Nix, M.F. Hutchinson, J.P. McMahon, and P.M. Fleming. 1988. Assessing representativeness of places for conservation reservation and heritage listing. *Environmental Management* **12**, 501–14.

Mackey, B.G., H.A. Nix, J. Stein, E. Cork, and F.T. Bullen. 1989. Assessing the representativeness of the Wet Tropics of Queensland World Heritage Property. *Biological Conservation* **50**, 279–303.

Mackey, B.G. and R. Sims. 1993. A climatic analysis of selected boreal tree species and potential responses to global climate change. *World Resources Review* **5**, 469–87.

Margules, C.R. and R.L. Pressey, 2000. Systematic conservation planning. *Nature* **405**, 243–53.

McIntyre, S. and R.J. Hobbs. 1999. A framework for conceptualizing human impacts on landscapes and its relevance to management and research. *Conservation Biology* **13**, 1282–92.

Morton, S.R. 1990. The impact of European settlement on the vertebrate animals of arid Australia: a conceptual model. *Proceedings of the Ecological Society of Australia* **16**, 201–13.

Morton, S.R., D.M. Stafford Smith, M.H. Friedel, G.F. Griffin, and G. Pickup. 1995. The stewardship of arid Australia: ecology and landscape management. *Journal of Environmental Management* **43**, 195–218.

NASA. 2003. *The Moderate Resolution Imaging Spectroradiometer (MODIS)* (http://modis.gsfc.nasa.gov/).

Nix, H. A. 1974. Environmental control of breeding, post-breeding dispersal and migration of birds in the Australian region. Pages 272–305 in H.J. Frith and J.A. Calaby (eds.). *Proceedings of the XVI International Ornithological Congress*. Canberra: Australian Academy of Sciences.

Nix, H.A., D.P. Faith, M.F. Hutchinson, *et al.* 2001. *The Biorap Toolbox: A National Study of Biodiversity Assessment and Planning for Papua New Guinea*. Canberra: The Centre for Resource and Environmental Studies, The Australian National University.

NLWRA. 2002. *Australian Terrestrial Biodiversity Assessment*. Canberra: National Land and Water Resources Audit.

Noss, R.F., C. Carroll, K. Vance-Borland, and G. Wuerthner. 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the Greater Yellowstone Ecosystem. *Conservation Biology* **16**, 895–908.

Noss, R.F. and A.Y. Cooperrider. 1994. *Saving Nature's Legacy*. Washington, DC: Island Press.

NTK. 2003. *Developing a National Biodiversity and Climate Change Action Plan*. National Task Group on the Management of Climate Change Impacts on Biodiversity. Canberra: Australian Government, Department of Environment and Heritage.

Oksanen, L. and T. Oksanen. 2000. The logic and realism of the hypothesis of exploitation ecosystems. *The American Naturalist* **155**, 703–23.

Palmer, C. and J.C.Z. Woinarski. 1999. Seasonal roosts and foraging movements of the black flying fox *Pteropus alecto* in the Northern Territory: resource tracking in a landscape mosaic. *Wildlife Research* **26**, 823–38.

Paton, D.C., A.M. Prescott, R.J. Davies, and L.M. Heard. 2000. The distribution, status and threats to temperate woodlands in South Australia. Pages 57–85 in R.J. Hobbs and C.J. Yates (eds.). *Temperate Eucalypt Woodlands in Australia: Biology, Conservation, Management and Restoration*. Chipping Norton, NSW: Surrey Beatty & Sons.

Possingham, H.P., Andelman, S.J., Noon, B.R., Trombulak, S. and Pulliam, H.R. 2001. Making smart conservation decisions. Pages 225–44 in G. Orians and M. Soule (eds.). *Research Priorities for Conservation Biology*. Covelo: Island Press.

Possingham, H.P., I.R. Ball, and S. Andelman. 2000. Mathematical methods for identifying representative reserve networks. Pages 291–306 in S. Ferson and M. Burgman (eds.). *Quantitative Methods for Conservation Biology*. New York: Springer.

Price, O. and D.M.J.S. Bowman. 1994. Fire-stick forestry: a matrix model in support of skillful fire management of *Callitris intratropica R.T. Baker* by north Australian Aborigines. *Journal of Biogeography* **21**, 573–80.

Reid, J.R.W., A. Kerle, and S.R. Morton. 1993. *Uluru Fauna: the Distribution and Abundance of Vertebrate Fauna of Uluru (Ayers Rock – Mount Olga) National Park*. Canberra: N. T. ANPWS.

Roshier, D.A., P.H. Whetton, R.J. Allan, and A.I. Robertson. 2001. Distribution and persistence of temporary wetland habitats in arid Australia in relation to climate. *Austral Ecology* **26**, 371–84.

Russell-Smith, J., P.G. Ryan, D. Klessa, G. Waight, and R. Harwood. 1998. Fire regimes, fire-sensitive vegetation and fire management of the sandstone Arnhem Plateau, monsoonal northern Australia. *Journal of Applied Ecology* **35**, 829–46.

Saunders, D.A. and R.J. Hobbs (eds.). 1991. *Nature Conservation 2: The Role of Corridors*. Chipping Norton, NSW: Surrey Beatty and Sons.

SEAC. 1996. *Australia: State of the Environment 1996*. Collingwood: CSIRO Publications. (http://www.deh.gov.au/soe/).

Shapcott, A. 2000. Conservation and genetics in the fragmented monsoon rainforest in the Northern Territory, Australia: a case study of three frugivore dispersed species. *Australian Journal of Botany* **48**, 397–407.

Singleton, P.H., W. Gaines, and J.F. Lehmkuhl. 2001. Using weighted distance and least-cost corridor analysis to evaluate regional-scale large carnivore habitat connectivity in Washington. Pages 583–94 in *The Proceedings of the International Conference on Ecology and Transportation, Keystone, CO. September 24–27*.

Soulé, M.E., J. Estes, J. Berger, and C. Martinez del Rio. 2003. Ecological effectiveness: conservation goals for interactive species. *Conservation Biology* **17**, 1238–50.

Soulé, M.E. and R.F. Noss. 1998. Rewilding and biodiversity: Complementary goals for continental conservation. *Wild Earth* **8**, 18–28.

Soulé, M.E. and J. Terborgh (eds.). 1999. *Continental Conservation: Scientific Foundations of Regional Reserve Networks*. Washington DC: Island Press.

Stafford Smith, D.M. and S.R. Morton. 1990. A framework for the ecology of arid Australia. *Journal of Arid Environment* **18**, 225–78.

Starr, B.J., R.J. Wasson, and G. Caitcheon. 1999. *Soil Erosion, Phosphorous and Dryland Salinity in the Upper Murrumbidgee: Past Changes and Current Findings*. "Pine Gully," Wagga, NSW, Australia: Murrumbidgee Catchment Management Committee.

Terborgh, J., J.A. Estes, P.C. Paquet, *et al.* 1999. Role of top carnivores in regulating terrestrial ecosystems. Pages 39–64 in M.E. Soulé and J. Terborgh (eds.). *Continental Conservation: Design and Management Principles for Long-term, Regional Conservation Networks*. Washington DC: Island Press.

Thomas, C.D., A. Cameron, R.E. Green, *et al.* 2004. Extinction risk from climate change. *Nature* **427**, 145–8.

Vertessey, R.A., R. Benyon, and S. Haydon. 1994. Melbourne's forest catchment: effect of age on water yield. *Water* **21**, 17–20

Whelan, R.J. 1995. *The Ecology of Fire*. Cambridge: Cambridge University Press.

Williams, J.E. and A.M. Gill. 1995. *The Impact of Fire Regimes on Native Forests in Eastern NSW*. Sydney: NSW National Parks and Wildlife Service.

Williams, J.E., R.J. Whelan, and A.M. Gill. 1994. Fire and environmental heterogeneity in southern temperate forest ecosystems: implications for management. *Australian Journal of Botany* **42**, 125–37.

Woinarski, J., P. Whitehead, D. Bowman, and J. Russell-Smith. 1992. Conservation of mobile species in a variable environment: the problem of reserve design in the Northern Territory, Australia. *Global Ecology and Biogeography Letters* **2**, 1–10.

Woinarski, J.C.Z., D.J. Milne, and G. Wanganeen. 2001. Changes in mammal populations in relatively intact landscapes of Kakadu National Park, Northern Territory, Australia. *Austral Ecology* **26**, 360–70.

World Resources Institute. 2001. *World Resources 2000–2001. People and Ecosystems*. United Nations Development Programme, United Nations Environment Programme, World Bank, and World Resources Institute.

Yibarbuk, D., P. J. Whitehead, J. Russell-Smith, *et al.* 2001. Fire ecology and Aboriginal land management in central Arnhem Land, northern Australia: a tradition of ecosystem management. *Journal of Biogeography* **28**, 325–44.

DAVID BOWMAN

12

Using landscape ecology to make sense of Australia's last frontier

12.1 Introduction

Just as the nineteenth century was a period of great biological discovery, driven by exploration and worldwide expansion of Western culture, there is no doubt that the dramatic global environment changes, driven by exploitation and pollution of the biosphere, will characterize the twenty-first century. A spin-off of the expansion of industrial civilization, that is driving the planetary environmental crisis, is the development and widespread availability of powerful digital technologies, such as geographic information systems, global positioning systems, digital aerial photography, and satellite imagery. These technologies provide unique insights into the rate and scale of environmental disturbances at the landscape-scale, which in aggregate drive global change. Natural resource managers and decision-makers tasked to achieve ecological sustainability necessarily focus on the landscape scale. Let us call the science that examines the ecological interaction between humans and landscapes *landscape ecology* (Naveh and Lieberman 1984). This discipline has the advantage of building on numerous other disciplines, including pure and applied physical and biological sciences and the more ambiguous, nuanced, and subtler fields in the humanities that have a stake in landscapes, including anthropology, environmental history, and various themes of human geography (Head 2001). Such a polyglot and young science is inherently vulnerable to bouts of introspection and anxiety about the conceptual bounds of the discipline and its philosophical roots (Wu and Hobbs 2002). I submit that the strength and utility of the transdisciplinary perspectives for making sense of and responding to global change is provided by landscape ecology. Further, these strengths are most apparent on a development frontier, such as the Australian monsoon

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tropics. Here, there is a remarkable juxtaposition of a technologically advanced society and a traditional indigenous population, set within a great expanse of minimally developed and biologically diverse tropical and arid landscapes. This globally unique situation also contrasts with the southern half of the Australian continent that has been transformed by European colonization in a mere 200 years of settlement.

In this chapter I draw on my experience in working on the north Australian frontier, reflecting on the potential of landscape ecology to contribute to the quest for sustainability in a time of tremendous environmental change. In such a culturally contested and rapidly changing region, the holistic and integrative approaches of landscape ecology are clearly apparent. So too is the power of story telling. Indeed, I explore these ideas by telling a number of "stories" about northern Australia, and my impressions of the practice of landscape ecology.

12.2 The north Australian frontier

The extraordinary diversity of endemic plant and animal species that so astonished and perplexed European colonists and observers such as Charles Darwin has rendered the keyword "Australia" synonymous with biological exceptions to the global rule, or at least the "normal" northern hemisphere rule. It is not surprising, therefore, that the study of Australian ecology has developed some independent traditions relative to global trends in ecology and evolution (Attiwill and Wilson 2003). The sense of studying the "exceptions" is amplified for those who work in northern Australia, a vast tropical frontier which, while sharing many biological similarities with southern Australia, has a number of salient physical and cultural features that make it different from the rest of the continent (Haynes *et al*. 1991). First, the climate of the north is controlled by the Australian summer monsoon. Relative to southern Australia, the north has back-to-front seasons: during the austral summer months (December to February), when the south of the continent regularly experiences high temperatures and oven-dry winds, the north has a hot, humid wet season characterized by week-long deluges that cause widespread flooding, and frequent tropical cyclones that wrack coastal regions. Conversely, during the austral winter months (June to August), the north enjoys a hot, rain-free austral "winter" while the southern days are short, chilly, and often wet.

In a mere 150 years there has been a remarkable confluence of a 40000 year-old culture with the Western tradition, associated with the last wave of colonization by the British Empire. In addition to perceived nineteenth-century geopolitical strategic imperatives, the economic drivers of northern settlement were exploitation of the endless landscapes by extensive cattle ranching and localized mining (Powell 1996). Infertile soils, labor shortages and isolation from markets stymied intensive agricultural development, so the northern Australian savannas experienced an insignificant degree of land clearing compared to southern Australia where agricultural development has seriously fragmented the native vegetation. Indeed, the north has the largest expanse of intact savannas of any region in the world, although this may change given developments in agricultural technologies and increasing global demand for food and fiber. Despite the social upheavals caused by European settlement, Aboriginal people that remain on their tribal lands have maintained one of the most ancient connections between humans and landscapes anywhere on Earth (Mulvaney and Kamminga 1999). It is increasingly clear that Aboriginal fire and game management has molded vegetation such that apparently "natural" landscapes have a profound "cultural" imprint (Yibarbuk *et al*. 2001).

For the first half of the twentieth century the north remained an exotic frontier, so far over the horizon that it was beyond serious consideration in national, let alone international, thinking. Sustained attacks by the Japanese during the Second World War, however, forced the Australian government to consider more seriously the future and potential of the north. The post-war period saw great investment by the Australian government in examining the economic potential of the vast "empty" landscapes using, for their time, advanced scientific and technological approaches, particularly aerial photographic mosaics. Indeed, these federally funded resource surveys prompted the development of "land system mapping," an often-overlooked, pioneering approach to "landscape ecology" (Christian and Stewart 1953). Land system mapping sought to characterize the edaphic, topographic, and biological resources as integrated mapping units.

In the last two decades of last century, the focus of landscape-scale research has shifted from exploring development potential to moderating the impact of developments and land management practices. There is a real risk that development will destroy the heterogeneity of the savanna habitat mosaic or disrupt the capacity of highly mobile wildlife assemblages to track resources in time and space (Bowman 1991, Woinarski *et al*. 1992). Examples of some of these research programs are the mapping of vegetation and fire activity across northern Australia (Fox *et al*. 2001, Russell-Smith *et al*. 2004), and the design of entire systems of conservation reserves using biological databases (Woinarski 1996, Woinarski *et al*. 1996, Parks and Wildlife Commission of the Northern Territory undated). These ambitious landscape-scale projects have triggered or provided important context for more focused research such as explaining why soils, landforms, and vegetation should be so closely coupled (e.g., Bowman and Minchin 1987), predicting the negative consequences of loss of naturally occurring rainforest isolates on other rainforest isolates (e.g., Price *et al*. 1995), or discerning the landscape pattern of Aboriginal fire usage (e.g., Bowman *et al.* 2004).

Clearly, the research priorities in the north contrast starkly with those undertaken in fragmented landscapes that characterize much of the developed world. The latter studies demand consideration of a finer spatial scale with an emphasis on corridors and fragments of habitats, and ecological restoration (Egan and Howell 2001, Liu and Taylor 2002). Nonetheless, there can be no doubt that quantitative landscape-scale analyses are a basic prerequisite for comprehending and formulating management of natural resources in a frontier like the north of Australia.

12.3 This is not a landscape

While the products of quantitative landscape ecology are of tremendous importance in framing and disciplining thinking about landscape, it is easy to overlook that they are at best a crude analog of an infinitely complex landscape. This devastatingly simple philosophical point was made by the surrealist painter Rene Magritte in this famous painting *The Treachery of Images*. The painting depicts a tobacco pipe, with a caption that reads "*Ceci n' est pas une pipe*" [this is not a pipe] (Foucault 1973). Magritte was making the point that the image is a representation of the thing, not the thing itself. Treating an abstraction as if it were a material thing is a philosophical fallacy known as reification.

Quantitative landscape ecologists routinely reify because it is not possible to providing a rigorous and unambiguous definition of landscape. This problem arises because of the fractal, multidimensional, and dynamic nature of landscapes and their ecological complexity and biological diversity. I suggest that the conceptual ambiguity and overwhelming complexity of "landscape" has resulted in two related tendencies in quantitative landscape ecology: the search for some absolute mathematical expression of landscape attributes – the so-called metrics – and the construction of deductive arguments about speculative "landscape processes."

12.4 The quadrat is dead

The search for metrics to mathematically describe "landscape" mirrors phytosociologists' quixotic quest in the second half of the twentieth century for the best numerical methodology to objectively define plant communities (Mueller-Dombois and Ellenberg 1974). In the course of this methodological development, the quadrat underwent a conceptual transformation from being a pragmatic device to help focus sampling effort to having an ill-defined "essence" that disconnected the observer from the reality and inherent complexity of vegetated landscapes. Great intellectual effort yielded rigorous sampling designs and numerical procedures to analyze quadrat data, and these undoubtedly revolutionized plant ecology. However, the outputs of the most sophisticated analyses (Jongman *et al*. 1995) remain what they always were – abstract and imperfect descriptions of vegetation based on small samples located in complex and dynamic landscapes. Similarly, there are no inherently right metrics; rather the choice of metric depends upon the purpose and context of the research.

12.5 Landscape models: but "there is no there there"

Another consequence of reification in quantitative landscape ecology is the overemphasis on deductive reasoning such as hypothetical models or flow diagrams of putative ecological processes. There is no doubt that such approaches are of great heuristic value. For example, recognition of the importance of localized fertile patches within tracts of infertile savanna has contributed to important insights into the mechanisms of widespread population declines and local extinctions of wildlife (Hobbs, in press). However, discussion of these models and idealized systems can overlook the fact that the components of these systems do not exist as discrete elements in the real world, or to use Gertrude Stein's famous phrase, "there is no there there." The operational definition of theoretical landscape elements is fraught, whether in the field or from remote-sensing data. A good example of this problem concerns the definition and delineation of the small naturally occurring fragments of "rainforest" that occur embedded in the north Australian savanna matrix. While there is agreement that such vegetation exists and is of great importance for the ecological function of the savanna matrix, there is no agreement as to how to define such vegetation (e.g., Lynch and Neldner 2000, Bowman 2001a). To avoid any pretence of absolute definitions and to sidestep a spiral of endless disputation, I suggest that landscape elements should be explicitly and operationally defined at the outset of a research program. This pragmatic approach requires one to take into account the environmental context, spatial scale, and purpose of the research. I believe that both theoretical and methodological development in quantitative landscape ecology must be literally grounded by comprehensive field checking and experimentation. An example of this approach was the recent calibration of four different methodologies to map fire scars using a landscape-scale field experiment (Bowman *et al*. 2003).

On a frontier, there is close engagement with landscapes and with land managers. Consequently, there is strong "selection pressure" to use quantitative techniques to document change rather than to invent, refine, and perfect techniques in isolation from pressing demands. Nonetheless, a real danger with such pragmatism is losing sight of the inherent value of theoretical research and becoming increasing intellectually isolated from debates, innovations, and developments of the rest of the discipline. The extraordinary opportunities to observe a rapidly changing environment combined with the tension between the pure and the applied aspects of landscape ecology makes the research on a frontier stimulating and challenging if researchers are adaptable. But such adaptability may take quantitative researchers into the qualitative realm.

12.6 Longing and belonging

Despite the intractable difficulties in neatly defining, quantifying, or even agreeing about the essential nature of "landscape," there can be no doubt that this concept is vitally important for land managers and the broader community. Discussion about "landscape" may act as a lightning rod for profound political and cultural debates about identity, place, and belonging. Both frontier and post-colonial societies typically have a keenly felt need for a sense of belonging to the landscapes they violently appropriated from indigenous people (Head 2000). Nature conservation and national-park movements signal a philosophical shift from the initial frontier mentality to a view that landscapes need to be cherished and preserved (Bonyhardy 2000). Aldo Leopold (1949), widely regarded as the father of the contemporary conservation movement, argued that sustainable settlement requires the development of a deep appreciation of where the settlers live. He argued that such environmental awareness involved an appreciation of geographical and historical contexts such as understanding how landscapes have evolved and how they change in response to seasonal cycles. For example, people need an appreciation of where landscapes fit in relationship to each other, from whence the winds blow, rivers flow, and migratory birds come and where they go. Leopold argued that the reciprocal interaction of land and people created a profound sense of belonging because humans and land had a closely shared history. He also realized that these interactions must be moderated by ethical restrictions on the scale and nature of resource usage.

Ironically, Leopold, like so many settlers, failed to grasp that such a holistic frame of landscape also underpinned the indigenous cultures that had been so dramatically and negatively transformed by settlement. For example, Australian Aborigines living on their "country" have an encyclopedic geographic and ecological knowledge, including complex mythologies about how landscapes were shaped and formed in the distant past during the so-called Dreamtime. Aboriginal people have profound physical and spiritual interconnections with landscapes that are formed and maintained through everyday use of natural resources such as hunting and gathering. Far from being "abstract," Aboriginal art often explicitly depicts complex practical and mythological knowledge of landscapes, but unlocking this knowledge requires an appropriate cultural frame (Watson 1993).

There can be no doubt that quantitative landscape ecology can play a pivotal role in the development of a sense of connectedness between settlers and their adopted landscapes. Furthermore, the discipline can be used to moderate the impacts of settlement. For example, policy-makers and land managers can anticipate areas of resource conflict and identify the geographic bounds of landscapes that are vulnerable to overexploitation or that have significant cultural and ecological values. But how does quantitative landscape ecology meet the challenge of communicating the great complexity, diversity, and dynamics of landscapes? If the discipline's outputs are often ambiguous, uncertain, and implicitly steeped in human values and perceptions, how can it meet the expectations of technocrats and policy-makers that want neat, black-and-white "answers"?

12.7 Tell me a story

Environmental historians have demonstrably succeeded in meeting some of the expectations, or at least capturing the attention, of land managers and policy-makers, notwithstanding limited field experience and a humanities background. How is it that they have succeeded in being so attuned to the public imagination? The answer, I believe, is that they have mastered the craft of telling stories about nature (Bowman 2001b and references therein). A good example of this genre is Stephen Pyne's (1998) essay about the dramatic shift in perceptions of the Grand Canyon by European colonists. In 1540 the Spanish explorers, who were the first Europeans to encounter the Canyon, dismissed it as hostile and worthless and nothing more than a geographic obstruction, yet 400 years later it is regarded as a natural wonder and part of the core of modern American identity.

According to the American environmental historian William Cronon (1992), environmental histories are nonfictional narratives that are steeped in human values and therefore interesting to people. These narratives are grounded in ecological fact but not limited by them. Environmental historians are unafraid of making much with little by breathing human interest into disjointed and incomplete facts. Environmental activists have also understood the power of stories and emotion in their campaigns to sway public opinion. This modus operandi often destabilizes and perplexes natural scientists who are uncomfortable with acknowledging and integrating human values into their thinking and quantitative analyses because it confounds the core principle of scientific "objectivity." This restrictive worldview, however, can be a great handicap in building a constituency to support research and to transform research outputs into real-world outcomes. This is particularly so for the media that is interested in presenting a "story" told by animated, interesting individuals.

Just as landscapes are open to many interpretations, so too are stories about landscapes. For example, the gross generalization that Australia is an arid, isolated, infertile chunk of Gondwana has been used to bolster claims to restrict the growth of human populations (Flannery 1994). However, these geographic facts have also been used to argue that "outback" Australia is a prime location for the storage of nuclear waste (Bowman 2004). Another danger is that a particular story can become orthodoxy because it is psychologically satisfying or fits a particular ascendant ideology, even though it may be based on slender evidence. It may then be difficult to subject the story to critical analysis, and it may be misused to achieve political ends. A good example of the power of stories relates to the massive clearing of native vegetation in central Queensland, Australia, where sketchy evidence of increases in the density of tree cover in some landscapes was used to justify wholesale destruction of native vegetation (see Bowman 2001b and references therein).

Landscape ecologists have the ability and the tools to record new data and to both critically assess the hypotheses that underpin existing stories and to create entirely new stories about landscapes. A concrete example is the use of aerial photography to detect the spatial extent and temporal pattern of landscape-scale vegetation cover change (e.g., Fensham and Fairfax 2003). This research has been used to bolster the hypothesis proposed by Fensham (2000) that landscape-scale changes in tree cover are better explained by drought cycles than by overgrazing or changed fire regimes. This "story" has far-reaching implications for attitudes and policy formulation about native-vegetation clearance.

Landscape ecologists often unwittingly tell stories about landscapes in their academic writings. If they are to influence land management, they must become comfortable with the power of story telling as a tool to reach a broader audience and encode information that can be readily comprehended by managers and policy-makers. Rather than making specific predictions about what will happen, Cronon (1993) suggested it is wiser to "offer *parables* about how to interpret what *may* happen" [original emphasis]. The stories landscape ecologists tell should not disguise the uncertainty that surrounds them, and their stories should be continually reevaluated in the light of both new research and practical experience. Indeed, environmental histories can be seen as hypotheses for active adaptive management systems (Walters and Holling 1990). The tools of landscape ecology can be employed to examine such hypotheses in a more or less rigorous – or at least quantitative – way (Whitehead *et al.* 2002).

12.8 Unexpected insights: confessions of an empiricist

As outlined in the following two vignettes, I have gained unexpected and, for me at least, profound insights in the course of undertaking field research for quantitative landscape ecology programs in northern Australia. These insights may be more important than the quantitative papers that I have produced, because they collapse my academic research into a compressed, emotionally charged, and insightful story that lives within me and continues to animate me. While the power of stories is self-evident for journalists, educators, advertisers, politicians, and scholars with a humanities background, their importance has only recently being explicitly acknowledged by applied scientists. For example, a recent paper in the *British Medical Journal* used "stories" to illustrate the relevance of complexity science to medical practitioners (Plsek and Greenhalgh 2001).

12.8.1 Shooting sacred buffalo

The original purpose for my field research in central Arnhem Land was to study the landscape-scale pattern of Aboriginal fire use (Bowman *et al*. 2004). My research agenda was opportunistically modified to include a study of buffalo (*Bubalus bubalis*) hunting (Bowman and Robinson 2002). This addition came about because I quickly realized that buffalo hunting was a potent social and (in nonmarket terms) economically productive activity that could bridge the cultural divide between me (a scientist) and the Aboriginal owners of the land upon which I was working. Buffalo hunting enabled us to experience the same landscape together. On one hunting trip, after killing a buffalo, I sat resting under a shady tree in a dry creek bed with my linguist collaborator Murray Garde and my Aboriginal informant and friend Joshua Rostrum. At my request, Murray asked Joshua, in Guene language, "What is a buffalo?" He was confidently told "the rainbow serpent." In that instant I grasped the ongoing complex and confusing relationship between humans and feral buffalo in northern Australia.

12.8.2 This is my land

I am currently examining changes in the distribution of mulga shrublands (*Acacia anuera*) and spinifex hummock grassland (*Triodia* sp.) in the Tanami Desert region of northern Australia. These vegetation types are powerful bio-indicators of landscape change in response to Aboriginal fire management practices. Long-term changes are being assessed through the examination of carbon isotope composition ($\delta^{13}C$) and radiometric dating (^{14}C) of soil organic matter (Witt, 2002). Medium-term changes and a landscape-scale context are being provided by a Geographic Information System (GIS) analysis of fire, land management, and vegetation community boundary changes using satellite and historical aerial photography. Shorter-term changes are being described using transects across vegetation boundaries stratified in respect of fire history, land tenure, and soil type. Recently I accompanied a group of government pastoral officers to arrange for permission to continue working on a pastoral property. We had chosen a bad day – the pastoralists had just received word that, due to a family company dispute, their lease had been sold and they were going to be evicted. The owner had been raised on the property and their children born there. The family's distress was palpable; they were going to lose more than an enterprise because their identity was strongly tied to that landscape. On the way to the pastoral property we had refueled at an Aboriginal community. The evidence of chronic social dysfunction was depressing and confronting. Momentarily, I held these two confronting scenes in my mind – each in their own way demonstrated the potent nexus between land and human identity. At that moment I comprehended the human dimension of the landscape change I was quantifying.

12.9 Conclusion

The great advantage of working on a frontier is that everything is new and change is the norm. Traditions in scientific thinking can only be made, not broken. My thinking about landscapes has been grounded by two decades of fieldwork in the great, uncompromising landscapes of northern Australia. This experience has shaped my attitudes towards landscape ecology in ways that practitioners from beyond the frontier might consider iconoclastic. I have learnt that there is no single approach to studying landscape ecology. Rather, context and purpose should determine choice of methodology, and an historical perspective is critical. Meeting the challenge of comprehending landscapes and formulating ecologically sustainable land management practices demands creativity and collaboration with scholars from a diversity of fields. In a frontier setting, where the consequences of the historical dispossession of indigenous people of their land by settlers suffuses all political discourse, it is impossible and counter-productive to deny that human values profoundly influence the way landscape is conceptualized and experienced. The fundamental importance of human values in landscape ecology favors the framing of stories or the formulation of parables rather than encouraging the quixotic quest for an "objective" representation of landscape. I have come to accept that it is impossible to ever comprehensively describe and truly understand landscape: the only truth is the landscape itself.

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References

Attiwill, P.M. and B. Wilson. 2003. *Ecology: An Australian Perspective*. Oxford: Oxford University Press.

Bonyhardy, T. 2000. *The Colonial Earth*. Melbourne: The Miegunyah Press.

Bowman, D.M.J.S. 1991. How short do you cut the string? Biogeography, development and conservation in northern Australia. *Global Ecology and Biogeography Letters* **1**, 2–4.

Bowman, D.M.J.S. 2001a. Future eating and country keeping: what role has environmental history in the management of biodiversity? *Journal of Biogeography* **28**, 549–64.

Bowman, D.M.J.S. 2001b. On the elusive definition of "Australian rainforest": response to Lynch and Neldner (2000). *Australian Journal of Botany* **49**, 785–7.

Bowman, D.M.J.S. 2004. Painful choices. *Nature Australia* **28**, 84.

Bowman, D.M.J.S. and P.R. Minchin. 1987. Environmental relationships of woody vegetation patterns in the Australian monsoon tropics. *Australian Journal of Botany* **35**, 151–69.

Bowman, D.M.J.S. and C.J. Robinson. 2002. The getting of the Nganabbarru: observations and reflections on Aboriginal buffalo (*Bubalus bubalis*) hunting in Northern Australia. *Australian Geographer* **33**, 191–206.

Bowman, D.M.J.S., A. Walsh, and L.D. Prior. 2004. Landscape analysis of Aboriginal fire management in Central Arnhem Land, north Australia. *Journal of Biogeography* **31**, 207–23.

Bowman, D.M.J.S., Y. Zhang, A. Walsh, and R.J. Williams. 2003. Experimental comparison of four remote sensing techniques to map tropical savanna fire-scars using Landsat-TM imagery. *International Journal of Wildland Fire* **12**, 341–8.

Christian, C.S. and G.A. Stewart. 1953. *General Report on Survey of Katherine-Darwin Region, 1946*. CSIRO Land Research Series No. 1. Melbourne: Commonwealth Scientific and Industrial Research Organization.

Cronon, W. 1992. A place for stories: nature, history and narrative. *The Journal of American History* **78**, 1347–76.

Cronon, W. 1993. The uses of environmental history. *Environmental History Review* **17**, 1–22.

Egan, D. and E.A. Howell. 2001. *The Historical Ecology Handbook: A Restorationist' s Guide to Reference Ecosystems*. Washington, DC: Island Press.

Fensham, R.J. 2000. *Nature' s Bulldozer: Tree Dieback in the Savannas* (http://savanna.ntu.edu.au/publications/savanna_links14/tree_dieback.html).

Fensham, R.J. and R.J. Fairfax. 2003. Assessing woody vegetation cover change in northwest Australian savannas using aerial photography. *International Journal of Wildland Fire* **12**, 359–67.

Flannery, T.F. 1994. *The Future Eaters: An Ecological History of the Australasian Lands and People*. Sydney: Read Books.

Foucault, M. 1973. *This is Not a Pipe* (trans. James Harkness). Berkeley: University of California Press.

Fox, I.D., V.J. Neldner, G.W. Wilson, and P.J. Bannink. 2001. *The Vegetation of the Australian Tropical Savannas*. Brisbane: Environment Protection Agency.

Haynes, C.D., M.G. Ridpath, and M.A.J. Williams. 1991. *Monsoonal Australia: Landscapes, Ecology and Man in the Northern Lowlands*. Rotterdam: Balkema.

Head, L. 2000. *Second Nature: The History and Implications of Australia as Aboriginal Landscape (Space, Place, and Society)*. Syracuse: Syracuse University Press.

Head, L. 2001. *Cultural Landscapes and Environmental Change*. London: Arnold.

Hobbs, R. Landscapes, ecology and wildlife management in highly modified environments: an Australian perspective. *Wildlife Research* (in press).

Jongman, R.H.G., C.J.F. ter Braak, and O.F.R. van Tongeren. 1995. *Data Analysis in Community and Landscape Ecology*. Cambridge: Cambridge University Press.

Leopold, A. 1949. *A Sand County Almanac, and Sketches Here and There*. 1989 Commemorative Edition. New York: Oxford University Press.

Liu, J. and W.W. Taylor. 2002. *Integrating Landscape Ecology into Natural Resource Management*. Cambridge: Cambridge University Press.

Lynch, A.J.J. and V.J. Neldner. 2000. Problems of placing boundaries on ecological continua: options for a workable national rainforest definition in Australia. *Australian Journal of Botany* **48**, 511–30.

Mueller-Dombois, D. and H. Ellenberg. 1974. *Aims and Methods of Vegetation Ecology*. New York: John Wiley & Sons, Inc.

Mulvaney, J. and J. Kamminga. 1999. *Prehistory of Australia*. Sydney: Allen and Unwin.

Naveh, Z. and A.S. Lieberman. 1984. *Landscape Ecology: Theory and Application*. New York: Springer.

Parks and Wildlife Commission of the Northern Territory (undated). *Northern Territory Parks Master Plan: Towards a Secure Future.* Darwin: Parks and Wildlife Commission of the Northern Territory.

Plsek, P.E. and T. Greenhalgh. 2001. Complexity science: the challenge of complexity in health care. *British Medical Journal* **323**, 625–8.

Powell, A. 1996. *Far Country: A Short History of the Northern Territory*. Melbourne: Melbourne University Press.

Price, O., J.C.Z. Woinarski, D. Liddle, and J. Russell-Smith. 1995. Patterns of species composition and reserve design for a fragmented estate: monsoon rainforests in the Northern Territory, Australia. *Biological Conservation* **74**, 9–19.

Pyne, S.J. 1998. *How the Canyon Became Grand*. New York: Penguin Books.

Russell-Smith, J., C. Yates, A. Edwards, *et al.* 2004. Contemporary fire regimes of northern Australia, 1997–2001: change since Aboriginal occupancy, challenges for sustainable management. *International Journal of Wildland Fire* **12**, 283–97.

Walters, C.J. and C.S. Hollings. 1990. Large-scale management experiments and learning by doing. *Ecology* **71**, 2060–8.

Watson, H. 1993. Aboriginal: Australian Maps. Pages 28–36 in *Maps Are Territories: Science Is an Atlas*. Chicago: The University of Chicago Press.

Whitehead, P.J., J.C.Z. Woinarski, D. Franklin, and O. Price. 2002. Landscape ecology, wildlife management and conservation in northern Australia: linking policy, practice and capability in regional planning. Pages 227–59 in J. Bissonette and I. Storch (eds.). *Landscape Ecology and Resource Management: Linking Theory with Practice*. New York: Island Press.

Witt, B. 2002. Century-scale environmental reconstruction by using stable carbon isotopes: just one method from the big bag of tricks. *Australian Journal of Botany* **50**, 441–54.

Woinarski, J.C.Z. 1996. Application of a taxon priority system for conservation planning by selecting areas which are most distinct from environments already reserved. *Biological Conservation* **76**, 147–59.

Woinarski, J.C.Z., G. Connors, and B. Oliver. 1996. The reservation status of plant species and vegetation types in the Northern Territory. *Australian Journal of Botany* **44**, 673–89.

Woinarski, J.C.Z., P.J. Whitehead, D.M.J.S. Bowman, and J. Russell-Smith. 1992. Conservation of mobile species in a variable environment: the problem of reserve design in the Northern Territory, Australia. *Global Ecology and Biogeography Letters* **2**, 1–10.

Wu, J. and R. Hobbs. 2002. Key issues and research priorities in landscape ecology: an idiosyncratic synthesis. *Landscape Ecology* **17**, 355–65.

Yibarbuk, D.M., P.J. Whitehead, J. Russell-Smith, *et al.* 2001. Fire ecology and Aboriginal land management in central Arnhem Land, northern Australia: a tradition of ecosystem management. *Journal of Biogeography* **28**, 325–43.

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13

Transferring ecological knowledge to landscape planning: a design method for robust corridors

13.1 Introduction

There is still a big gap to cross between ecology and planning (Moss 2000, Opdam *et al.* 2002). This lack of integration is a problem in several ways. If a regional development plan projects a spatial pattern of ecosystems not sustaining the key ecological processes to serve the nature conservation objectives, it is by definition ecologically unsustainable. Moreover, for landscape ecology as an applied problem-solving science, its future and its justification (Moss 2000) is at stake, if landscape ecological knowledge is unable to provide a sound scientific basis for the planning of landscapes. In this chapter, we present an approach for the transfer of knowledge on population ecology to planning and design procedures. The method is based on two assumptions: regional stakeholders determine conservation targets as well as landscape design, and such decision-making is based on the principles of ecological sustainability. We developed this method in the context of the planning of robust corridors in the Netherlands.

Why should regional development plans be ecologically sustainable? Sustainable development is a widely accepted strategic framework in decision-making concerning land use now and in the future (IUCN 1992). It demands that landscape planning aims for "a condition of stability in physical and social systems, achieved by accommodating the needs of the present without compromising the ability of future generations to meet their needs" (WCED 1987, Ahern 2002). This implies that in decision-making about a future landscape a balance is achieved, in the short and long term and among ecological, cultural, and economic functions (Linehan and Gross 1998, Ahern 1999). It further implies that the spatial organization of the landscape sustains these functions in the long term. For species diversity, for example, this would imply that

Key Topics in Landscape Ecology, ed. J. Wu and R. Hobbs.

the pattern of ecosystems provides populations of species with the capacity to recover from local and regional disturbances.

However, ecological sustainability is not well developed in landscape planning for two reasons. The first one is that the explicit inclusion of ecological principles in landscape planning is quite a recent advancement (Ahern 2002). The second reason is that ecological sustainability has not been properly defined in a spatially explicit context. We will use the definition proposed by Opdam *et al.* (2005): a landscape is ecologically sustainable if: (1) the landscape structure supports the required ecological processes, so that the landscape can deliver its natural goods and services to present and future generations, (2) the landscape can change over time without losing its key resources, and (3) stakeholders are involved in decision-making about landscape functions and patterns. The first and second conditions require the understanding of the interdependence between landscape pattern and ecological and abiotic processes (Wu and Hobbs 2002). However, often landscape plans are not based on landscape ecological concepts (see Bergen Jensen *et al.* 2000, Steiner 2000, Tress *et al.* 2003). The third condition requires that nonecologists can handle information based on ecological knowledge, and this information is not presented as a standard recipe, but is open to decision-making and integration in a complex, multi-actor planning process.

The definition can be criticized for not being explicit about how many ecological services and goods are needed for present and future generations. We believe that defining ecosystem and biodiversity targets for a region is a context-dependent democratic activity, in which international as well as national conservation targets are considered in the regional context. Stakeholders are primarily responsible for setting ambition levels, with science in the role of providing a basis for decision-making. It is outside the scope of this chapter to discuss this issue, and we simplify it here by taking biodiversity as a resource that should be conserved in the landscape for future generations (Luck *et al.* 2003). This general notion can be transformed into the goal that populations of species are persistent in the long term. However, the required conservation effort increases with the amount of species in a planning area occuring in persistent populations (Opdam *et al.* 2005). This implies that planning for ecological sustainability always demands that a level of ambition is defined. We consider this as a political and societal question rather than a scientific one. This means that goal-setting for sustainable planning requires the involvement of stakeholders. Because conservation of biodiversity may not be compatible with other land-use functions, and requires money, goal-setting is part of a negotiation process. Ecological knowledge must be in a form that is suitable for such a process.

Our approach allows integration of landscape ecological knowledge into landscape planning and design. It entails three essential steps to bridge the gap between basic landscape ecological research and its application in landscape planning (also see Opdam *et al.* 2002, 2005):

- Step 1 – the translation of basic ecological knowledge of individuals and populations of a species into minimum levels of habitat quality, area, and configuration at a defined spatial scale.
- Step 2 – the integration of these conditions from single species into multi-species or ecosystem conditions.
- Step 3 – the transfer of that knowledge for use in the various steps of the planning process.

We focus on the total area and spatial distribution of habitat for a species in a region, assuming that abiotic fluxes and patterns allow for good-quality habitat, and that management of ecosystems is adequate. By this focus the chapter becomes less complicated, while it does not affect the message we want to get across. Moreover, we believe that in planning there is already quite some attention on the relation between ecosystems and abiotic conditions (Steiner 2000), whereas the significance of the spatial pattern of ecosystems to biodiversity is often neglected. A further simplification is that we neglect the response time of populations to changing ecosystem patterns. Though this is not realistic, this simplification does not alter the method we present, but it may alter the minimum area threshold of an ecosystem network.

We have developed our approach in the context of a case study: the recent introduction of a new concept in Dutch conservation policy: "robust corridors" (explained below). The aim of this chapter is to discuss the development of planning guidelines for effective corridors, based on the best available ecological knowledge, and the effective implementation of these guidelines in a complex multi-actor planning process. We will analyze the different steps in the implementation process from basic landscape ecology to on-the-ground application, and identify different types of knowledge and skills required in each phase. We are aware that no single "best method" exists for the implementation of landscape ecology in landscape planning. Strategies are, for instance, strongly dependent on the nature policy and planning traditions in various countries. However, we do think that our approach might be helpful to better understand the prerequisites for the integration of landscape ecology in planning.

13.2 Context of the case study

In the 1990s the Dutch government launched a far-reaching landscape ecological concept in nature policy: the National Ecological Network (NEN; MANFS 1990). The NEN was conceived as a structure of existing nature areas that was made more robust and cohesive by enlarging existing areas, developing new nature areas, and establishing local ecological corridors. The NEN was an answer to habitat loss and fragmentation, which were considered the prime causes for the loss of biodiversity. Habitat networks were thought to offer conditions for long-term conservation where individual areas were no longer large enough for persistent populations (Opdam *et al.* 1995, Hanski 1999, Hobbs 2002, Opdam 2002). Long-term survival of target species in habitat networks could be achieved when requirements were met concerning: total network size, habitat quality, network density, and connectivity of the network (so-called "network cohesion," Opdam *et al.* 2003). The biodiversity objectives of nature policy were defined in target ecosystem types and target species (Bal *et al.* 2001). In 2000, halfway through the implementation process, an evaluation took place to predict whether the NEN would indeed effectively protect these target species. It proved that the expected spatial cohesion after implementation of the NEN would still be insufficient for the sustainable conservation of the target species. Three main causes were identified:

(1) The nature areas were still too small. The autonomic process of obtaining land for nature development was only moderately successful in enlarging existing areas (Nature Policy Agency 1999).
(2) The connectivity was too low. The development of ecological corridors was too slow (Beentjes and Koopmans 2000) and about 50 percent of the planned corridors would be ineffective due to insufficient design (Bal and Reijnen 1997).
(3) Barriers caused by infrastructure were insufficiently solved (Reijnen *et al.* 2000).

A project in which both scientists and policy-makers participated resulted in a new landscape ecological concept, "robust corridors," which was chosen as the best solution for the lack of spatial cohesion (Pelk *et al.* 1999). Robust corridors connect the important nature regions in the Netherlands, over a distance of several tens of kilometers. Robust corridors consist of wide dispersal corridors and large new nature reserves. The concept was adopted by the national government (MANFS 2001), and it was decided to expand the NEN with supraregional robust corridors (Fig. 13.1). Subsequently, the governments of the 12 provinces were asked to explore the possibility of developing such robust corridors. In fact, this was the beginning of a negotiation process between the national government and the provincial governments about implementing the national policy at the provincial level. Because provinces were still in the process of implementing the original plan from 1991, and differed in the progress they had made and in the regional support by farmers, the new ambition of the national government was not received with enthusiasm everywhere. Moreover, the initial idea allocated unequal amounts of extra hectares to provinces. The

FIGURE 13.1
Robust corridors (dark lines) added to the National Ecological Network in the Netherlands to connect important nature regions (grey areas)

final outcome of this first step in the planning process was the decision by the national government about which robust corridors should be developed, the total area needed, and the distribution over the provinces.

We supported this negotiation process during the explorative phase with a framework for planning and design of ecologically effective robust corridors. To develop spatial conditions for robust corridors, we went through step 1 – "the translation of basic ecological knowledge of individuals and populations of a species into general spatial conditions of the landscape," and step 2 – "the integration of knowledge from single species to multi-species or ecosystem conditions." Then we transferred these guidelines into a planning and design

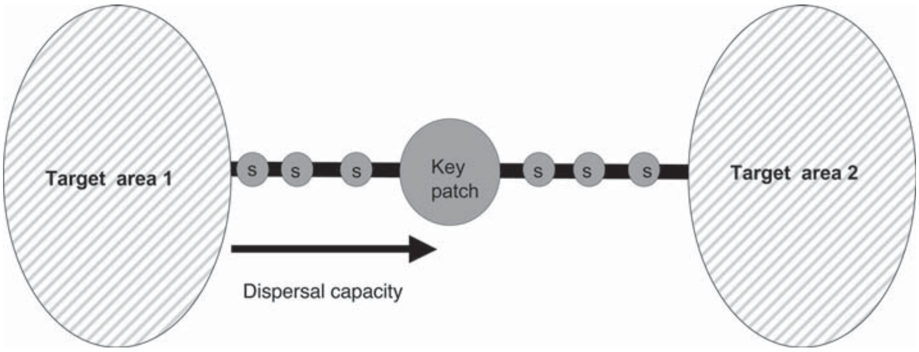


FIGURE 13.2
Translation of basic landscape ecological knowledge into spatial conditions for single species ecological corridors (step 1)

procedure to guide the feasibility studies carried out by the provinces. Here step 3 came into focus: "the transfer of knowledge for use in the various steps of the planning process." In the following sections we will describe the development of the guidelines for robust corridors and the implementation of design rules for robust corridors in the planning process.

13.3 The development of robust corridors and the implementation in the planning process

13.3.1 Step 1: the translation of basic landscape ecological knowledge into guidelines for single-species corridors

Robust corridors should facilitate the exchange of species between target areas through unsuitable landscapes over distances that will often exceed their dispersal capacity by far. Therefore the corridor consists of a combination of new habitat patches, where a species can establish and maintain a population, the so-called key patch, and measurements that will facilitate dispersal through an inhospitable matrix, so-called dispersal corridor and stepping stones (Fig. 13.2). A large body of basic landscape ecological research has formed the basis for guidelines for corridor conditions of single species. Empirical studies provide information on dispersal distances and whether a species needs special habitat elements (a dispersal corridor) to cross agricultural or urbanized landscapes (e.g., Bennett 1999, Rickets 2001, Ray *et al.* 2002, Tewksbury *et al.* 2002, Vos *et al.* 2002). Metapopulation models provide guidelines on the size of and distance between habitat patches, depending on the dispersal capacity and individual area requirements of target species (e.g., Verboom *et al.* 2001, Etienne *et al.* 2004, Ovaskainen and Hanski 2004,

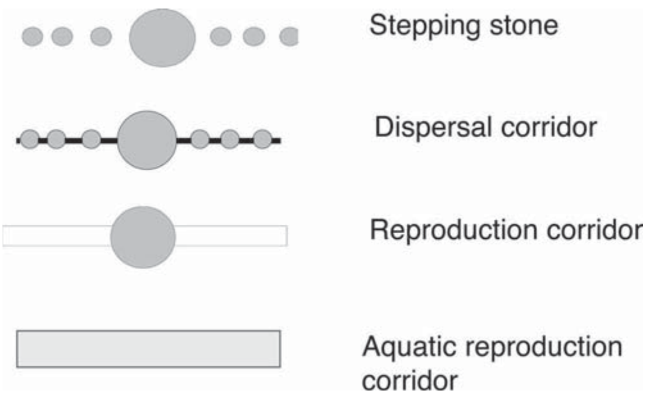


FIGURE 13.3
Four basic corridor types based on the dispersal capacity and mode of dispersal of the target species

Fahrig, Chapter 5, this volume). Rules for the required patch size of key patches were derived from metapopulation modeling and knowledge on individual area requirements of species (Verboom *et al.* 2001). Rules for the distance between habitat patches were based on knowledge about the dispersal capacity of species. Empirical research on movements of species in heterogeneous landscapes in combination with movement modeling provided knowledge of whether species avoid or prefer specific habitat types during dispersal (Vos *et al.* 2002). Based on species-specific dispersal characteristics, this knowledge was extrapolated into rules for dispersal corridors, measures at infrastructural barriers, and stepping stones (Fig. 13.2). Four basic corridor design patterns were developed, related to different modes of dispersal (Fig. 13.3).

13.3.2 Step 2: integration from single species to multi-species robust corridors

Empirical studies on the functioning of species in heterogeneous landscapes are often based on single species. Nature conservation goals are not. Thus landscape planners need integrated corridor guidelines, where requirements for single species are integrated into a multi-species design. The integration was achieved in the following way (see Box 1 for more details). Dutch nature policy has formulated explicit biodiversity aims in a list of target species that are representative for ecosystem types (Bal *et al.* 2001). A robust corridor that connects two target areas (for instance two marshlands) should facilitate the exchange of all (marshland) target species. As a first step towards integration for each ecosystem type the target species were grouped according to their dispersal capacity, dispersal mode, and individual area requirements (Box 13.1). Target species with similar requirements were combined in

Box 13.1. Integration from single species to multi-species robust corridors (step 2)

(1) Integration from target species to ecoprofile corridors

For each ecosystem type the target species are categorized for relevant ecological corridor requirements: dispersal mode (by air, land, or water), dispersal distance classes ($\leq$1 km, 1–3 km, 3–7 km, 15–15 km, and $>$35 km), and area requirements for a key patch ($\leq$0.1 km^2, 0.1–1 km^2, 1–5 km^2, 50–150 km^2). Target species with the same ecological corridor requirements are combined in one ecoprofile (Vos *et al.* 2001, Pouwels *et al*. 2002). The different ambition levels for robust corridors (see Table 13.1) are linked to the ecoprofiles by their dispersal capacity. For the lowest ambition level the robust corridor functions for species with a dispersal capacity larger that 15 km. The medium ambition level incorporates also less mobile species (dispersal capacity $>$3 km), while on the highest ambition level all ecoprofiles are included.

(2) Integration from ecoprofile corridors to a multi-species robust corridor

In the last integration step the ecoprofile corridors from a particular ecosystem type and ambition level are integrated into one robust corridor that is suitable for all ecoprofiles. In the example (Fig. 13.4) a robust corridor for a marsh ecosystem at the lowest ambition level consists of four ecoprofiles: "beaver," "great reed warbler," "otter" and "bittern." Rules for integration are:

(i) distance between patches is determined by the ecoprofile with the lowest dispersal capacity ("beaver"), and

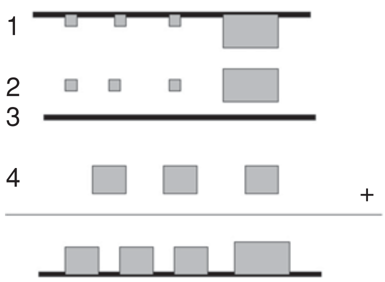
(ii) area of patches is determined by the species with the largest area requirements ("bittern"). This integration of ecoprofiles leads to an area reduction of 40 percent compared to the required area for all separate ecoprofile corridors.

(3) Finally a robust corridor might consist of several ecosystem types (see Fig. 13.4).

"ecoprofiles" (Vos *et al.* 2001, Pouwels *et al.* 2002). For instance, the amphibian species tree frog (*Hyla arborea*), the common spadefoot (*Pelobatus fuscus*), and the pool frog (*Rana lessonae*) have comparable habitat requirements, dispersal distances, and individual area requirements and therefore were classified into one ecoprofile. This resulted in a reduction of 398 target species to 138 ecoprofiles (Broekmeyer and Steingröver 2001). Subsequently the requirements of a list of ecoprofiles of the same ecosystem type were integrated into one robust corridor that fits requirements for all ecoprofiles. In this integration step the distance between key patches is determined by the ecoprofile with the smallest dispersal capacity, while the size of key patches is determined by the ecoprofile with the largest individual area requirements. As is explained in Box 13.1 (Fig. 13.4),

TABLE 13.1. *Robust corridors with low, medium, and high level of ecological ambition*

Ecological ambitions	Low ambition level (1)	Medium ambition level (2)	High ambition level (3)
Create sustainable networks on national scale	Very mobile species (dispersal capacity > 15 km), with large area requirements	–	+
Connect regional networks to increase biodiversity in all suitable habitat		Medium mobile species (dispersal capacity 3–15 km), with intermediate area requirements	+
Connect networks for the whole ecosystem: the spreading of risks			Species low mobility (dispersal capacity < 3 km), with small area requirements



Ecoprofiles	Distance between key patches	Area key patch, stepping stone	Dispersal corridor (width)
1 "Beaver"	20 km	300 ha, 30 ha	50 m
2 "Great reed warbler"	20 km	300 ha, 30 ha	-
3 "Otter"	50 km	not needed	50 m
4 "Bittern"	30 km	750 ha, 75 ha	-
Total	Robust corridor Marshland, ambition level 1, length 25 km		

FIGURE 13.4
Integrating ecoprofiles into one robust corridor for a marshland ecosystem on the lowest ambition level: suitable for relatively mobile species with large area requirements (dispersal capacity > 15 km)

an important reduction of the total required area is reached, by integrating all corridor-requirements of the ecoprofiles into one robust corridor.

13.3.3 Step 3: developing tools for the implementation of flexible design rules in the planning process

It was important that feasibility studies were carried out in the same way at the national and provincial levels, and that an analysis of cost-effectiveness produced a ranking of corridors. The provinces were asked to analyze the cost-effectiveness for different scenarios. In a handbook, examples of robust corridors were worked out (Broekmeyer and Steingröver 2001). For a flexible application of the design rules in the planning process a CD-ROM was produced providing information on all robust corridors per ecosystem type and per ambition level. Different scenarios could be generated by varying the ambition level or the number of ecosystem types that were incorporated in the corridor. During the feasibility studies, planners were searching for the most effective, best accepted, and economically most stable corridor options. To facilitate this negotiation process between the national and provincial governments, we decided to maximize flexibility in the planning and design guidelines. The following options allowed such flexibility: (1) defining the ambition level, (2) defining the ecosystem type(s) to be included, (3) finding the preferred location, (4) defining the sequence of corridor elements, and (5) combining other land-use functions.

13.3.3.1 Defining the ambition level

Conservation targets were divided in three ambition levels, depending on the number of species that should use the corridor (Table 13.1). At the lowest ambition level (i.e., level 1), robust corridors are created for only those species that require habitat networks on a national scale. These are the mobile species (dispersal capacity > 15 km) with very large area requirements, such as otter (*Lutra lutra*) and bittern (*Botaurus stellaris*). These species need enough cohesion between all large Dutch nature areas to create sustainable habitat networks. At ambition level 2, requirements for species that form viable population networks on a regional level are included in the corridor. These are moderately mobile species (dispersal capacity 3–15 km) with medium area requirements, such as grass snake (*Natrix natrix*) and blue throat (*Luscinia svecica*). Although these species are able to survive in networks on a regional level, the effectiveness of the NEN as a whole will improve by making exchange between these regions possible for moderately mobile species. Thus target species will be able to reach suitable habitat in regions where they would otherwise be missing, increasing species number per nature reserve. At the highest ambition level (3) the robust corridors will be suitable for the whole ecosystem and exchange between regions for all species should be possible. These corridors were believed necessary to allow species to respond to large-scale disturbances, such as climate change. When favorable habitat conditions shift as a result of climatic change, all species should be able to follow these changes within the Netherlands (Opdam and Wascher 2004). For this strategy to be effective, robust corridors are needed not only in the Netherlands but also in the whole European ecological network (so-called “Natura 2000” reserves; Jongman *et al.* 2003).

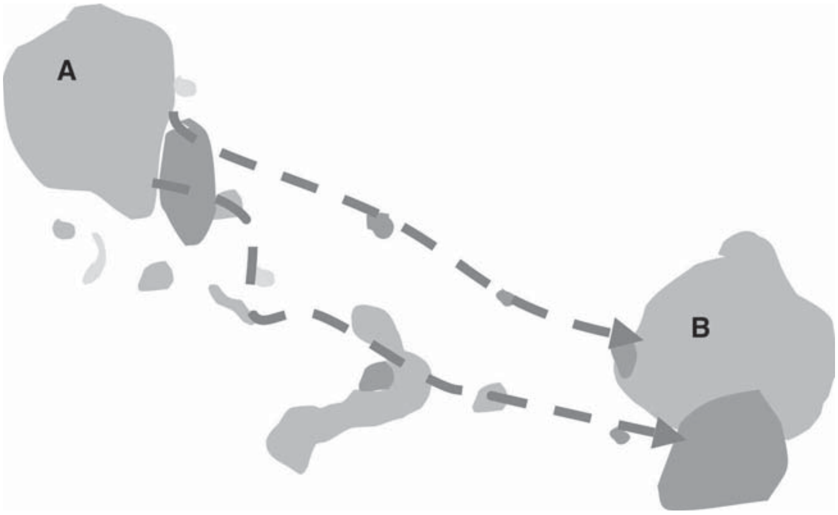


FIGURE 13.5
Two scenarios for a robust corridor between target areas A and B. The grey tones indicate different ecosystem types. The dashed lines are two scenarios for the location of the robust corridor. The top line would be the shortest route between A and B. However, this route would require large investments to develop new nature areas. The bottom line is longer, but incorporates more existing nature areas

13.3.3.2 Defining the number of ecosystem types in the robust corridor

Natural regions to be connected by robust corridors often consist of different ecosystem types, arranged in mosaics of woods, marshes, dry heath, moors, and grasslands. Therefore a robust corridor design should consider all ecosystem types present in a region. For instance, if one wants to enhance exchange of forest species over large distances, one would need a forested corridor and create new forested nature reserves. For heath species, the corridor needs to be extended with a heath corridor, and new heath areas, etc. One may, however, decide to develop the corridor zone for only one or two of the ecosystem types present, which would require less area and thus be easier to accomplish.

13.3.3.3 Finding the preferred location

Because the CD-ROM linked the scenarios directly to the number of target species that would benefit from the corridor, the potential benefit between scenarios could be compared. The best location for the corridor was determined by analyzing the cost-effectiveness of different options. The trade-off was compared between the shortest route between target areas and the route that incorporated most of the existing nature areas or encountered least barriers (Fig. 13.5). The optimal allocation was found by producing alternative trajectories of the corridor, and choosing between the options can be based on costs (amount of additional area required, number of barriers to be crossed) as opposed to the number of species that will benefit from the corridor.

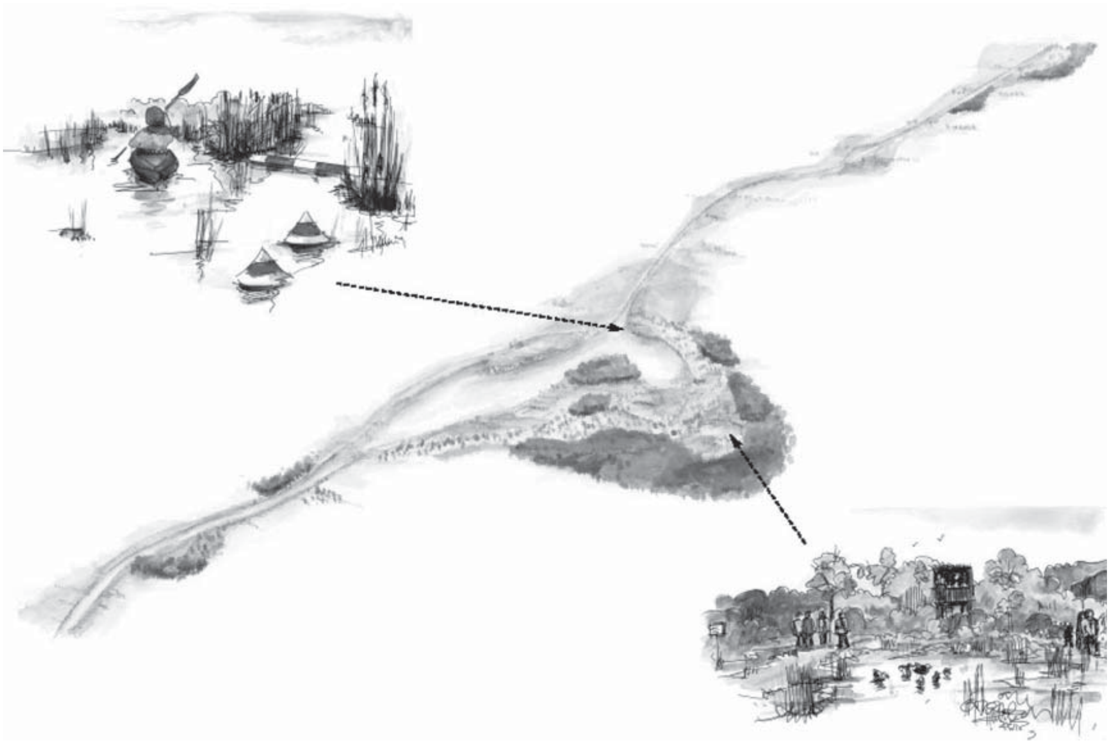


FIGURE 13.6
Visualization of function combinations. A robust marshland corridor can be combined, to some extend, with recreation (artist impression by Karel Hulstein; step 3). The illustration presents the robust corridor, consisting of a marshland dispersal-corridor and new marshland areas. The target areas that are to be connected by the robust corridor are not in the illustration (from Broekmeyer and Steingröver 2001)

13.3.3.4 Defining the sequence of corridor elements
Guidelines were incorporated as to how the sequence of corridor elements could be altered without losing the ecological effectiveness. By creating room for different design options within one ambition level, we intended to create flexibility in the design process, and thereby to increase the probability that a proper design could be negotiated between parties.

13.3.3.5 Combining other functions
Finally the handbook discusses the possibilities to combine the ecological function of the corridor with other functions of the landscape: water management, agriculture, and recreation. For instance, when recreation is stimulated by these corridors, this will be beneficial for the regional economy and contribute to gaining support for the plans (Fig. 13.6). However, recreation is also considered as a potential disturbance for animal populations, with the effect of

decreasing the carrying capacity of the habitat. We reviewed existing knowledge on the impact of recreation on species, (e.g., Hill *et al.* 1997, Henkens 1998, Miller *et al.* 1998, Vos *et al.* 2003) and filled knowledge gaps with best expert guesses on disturbance distances. This knowledge was summarized in a series of design principles for recreation facilities, including minimum distances and multiplying factors for minimum area to compensate for a potential decrease in habitat quality. Similar procedures were followed for the integration of particular forms of agriculture, cultural landscapes values, and water retention within these corridors.

13.4 Discussion

13.4.1 Contribution to key issues

Wu and Hobbs (2002) proposed six key issues of landscape ecology. In this chapter, we have mainly addressed three issues: integration between basic research and application, transdisciplinarity, and cooperation with policy-makers and stakeholders. Following Wu's and Hobbs' claim that effective communication requires willingness, desire, and commitment on the landscape ecologists' part, we have developed an interactive design approach with two main achievements: it links the required ecological functioning to the selected conservation goal, and it offers opportunities for decision-making. Therefore, it allows a context-oriented design of corridors, decisions by nonecologists that are still ecologically sustainable, and room for negotiation between parties in a planning and design process. We have integrated the three key issues, thereby facilitating interactive sustainable planning. Being "interactive" is important because the ecological researcher is not the decision-maker: his or her role is clarifying the boundary conditions for landscape patterns given a target chosen by stakeholders and policy-makers. The researcher also develops methods to facilitate the setting of feasible (nature conservation) targets. In addition to what we have developed in our case study, such methods may include the determination of ecosystem functions to be ensured in the region for future generations. In that role the researcher demarcates the playing field for the game the stakeholders have chosen, and warns where boundaries are crossed. If during the planning process it turns out that the required conditions are not attainable, the researcher facilitates the reformulation of goals. Methods like the design approach discussed here can provide the decision process with a scientific foundation, because they make the planning process systematic, transparent, and reproducible. Such methods can be peer-reviewed and discussed in scientific meetings, and thereby become acceptable for society as a general basis for decision-making.

Critical to this role of the researcher is that the method allows the actors to move and switch between alternatives in the negotiation process. As Theobald *et al.* (2000) pointed out, "Planning is a process that uses scientific data, but ultimately depends on the expression of human values." The actors explore the domain of sustainable options to find the solution that best fit their preferences. The best landscape design is not found on the basis of ecological criteria only. It may not be the cheapest option either, but the one with the best balance among ecological, social, and economic profits. It is a compromise rather than an optimal design from the point of view of ecological criteria (Haines-Young 2000). We have argued in this chapter that the compromise should not be found in accepting ecologically less sustainable conditions, but in lowering the ambition level of the objectives, requiring less area or a more feasible spatial pattern for the corridor.

Our approach does not use formal optimization methods (e.g., Cabeza and Moilanen 2001, McDonnell *et al.* 2002, Rothley 2002, Hof and Flather, Chapter 8, this volume). We believe that such methods are useful for generating a series of theoretical scenarios for the location of nature reserves, which could be useful in landscapes with a much lower intensity of current land use. The practical significance of such research increases if they could generate the domain of sustainable configurations, rather than the ecologically best solution, and if they could be made suitable for planning with interactive stakeholder involvement.

13.4.2 Further development of the corridor design method

13.4.2.1 Step 1: translating basic species ecology into spatial conditions

Future landscape ecological research will enhance our understanding of the functioning of species in fragmented landscapes. This will provide new insights about sustainable spatial conditions for species survival and consequently influence the design rules for robust corridors. Thus it is important that the structure of design rules is flexible, so that new knowledge can be incorporated and distributed easily (for instance by CD-ROM). As the implementation of the corridors in the landscape will be a long-term process, there is time to incorporate improved design rules. On the other hand, monitoring the effectiveness of the robust corridors might also generate basic knowledge on species functioning. As Golly and Bellot (1991) put it, "Landscape plans are actually hypotheses of how a proposed landscape structure will influence landscape processes." Thus the implementation of robust corridors in the landscape generates landscape experiments that in themselves form the basis for new knowledge on species functioning. Study questions, for instance, are whether the corridors are actually used by the target species and whether species presence in the target areas is increasing. In particular, monitoring the (genetic) species diversity in the target areas before and after implementation will generate strong evidence of its effectiveness.

13.4.2.2 *Step 2: knowledge integration*

Species-specific knowledge is not effective in decision-making by stakeholders who cannot handle the large variation in spatial scales and habitat requirements. We proposed a method in which species data were integrated in a framework for landscape design, and in which we tried to balance between loss of detail and necessary simplification and generalization. We combined corridor requirements for species that have roughly similar reactions to scale and configuration of habitat pattern into so-called ecoprofiles (Vos *et al.* 2001). An alternative approach is to identify the most critical species per landscape characteristic, for instance, area-limited and dispersal-limited species (Lambeck 1999). The integration problem has been tackled in various ways (e.g., Lambeck and Hobbs 2002), including the introduction of focal or umbrella species (Cox *et al.* 1994, Noss and Cooperrider 1994) that are expected to provide protection for other species. Knowledge integration is dependent on sound species-specific knowledge that comes from basic studies. Also, the development of integrative frameworks needs specific research input and the development of new approaches. Too often integrative methods are not published in scientific papers. This is detrimental to the development of conservation planning as a science because, when methods are not made transparent and repeatable, the effectiveness cannot be compared.

A relatively new challenge for basic landscape ecology is the development of spatial conditions for multifunctional ecosystem networks. In our approach, we gave some indications based on expert judgment on whether robust corridors could be combined with recreation, water management, and agriculture. However, the level of quantitative knowledge on the interrelationship between nature and other functions is quite poor. If ecological functions are impaired by other functions, how can ecosystem networks be designed to compensate for that loss? For instance, it is known that recreation pressure lowers the nest density and reproduction success for bird species (Yalden and Yalden 1990). Incorporating this decline in habitat quality will require larger habitat networks for viable bird populations (Vos *et al.* 2003). We urgently need more quantitative studies on the interaction between land-use functions allocated within ecosystem networks.

13.4.2.3 *Step 3: flexible design rules*

Flexibility is important if ecological structures are to be inserted into a specific multifunctional landscape context. End users of the corridor design approach could vary the ambition level by varying the type of ecosystems to be included in the corridor, as well as the number of species for which the corridor could serve as a functional connection. These choices have consequences for the amount of space and the abiotic conditions required. The designer can also shift the chain of corridor elements in the landscape to find the most appropriate locations with the least amount of area to be acquired. Then, as an additional degree of freedom, we built in the possibility to shift the order of elements, to optimize the fit of the corridor at the local level. Finally, we gave guidelines to use the corridor for water storage and recreation, including consequences for the design and the area required for ecological functioning. We hypothesize that such degrees of flexibility in ecological design rules increase the effectiveness of ecological knowledge in improving the ecological sustainability of landscape development. This aspect of the relation between ecology and planning deserves much more research, including formal testing of the hypothesis.

13.4.3 Impact on the planning process

We have shown how we were actively involved in the explorative phase of planning robust corridors. But how has our method affected the planning process? As Theobald *et al.* (2000) pointed out, the ultimate question is how ecological information has altered the decision-making process. We have not been able to measure this effect for our method, because we did not attempt any systematic comparison with other planning approaches. We do know, however, that the 12 provinces involved in the explorative studies all worked with the handbook and CD-ROM. Possibly because this converged their thinking and because the corridors were crossing provincial borders, we witnessed, for the first time in the ten-year history of the National Ecological Network, that all the provinces really worked together on common goals. They also readily accepted the priority ranking of the proposed corridors by the national government. We assume that our method and the role we played in facilitating the goal-setting improved the decision-making process and the acceptance of the new concept by the provinces. We were asked to evaluate whether in these explorative studies the handbook was appropriately applied. Although most users were positive about the value of the handbook, we found big differences in the interpretation of the steps of the method, possibly partly attributable to the very short time in which the explorations had to be made. We suspect that some users had insufficient basic knowledge of ecological processes, and no time to acquire all the background information presented in the handbook. We take this as an indication that the design tool still needs further adjustments, based on the experiences of users. This can only be achieved by a profound understanding of the planning process, and requires the interactive participation of landscape ecologists in landscape planning.

References

Ahern, J. 1999. Spatial concepts: planning strategies and future scenarios: a framework method for integrating landscape ecology and landscape planning. Pages 175–201 in J. Klopatek and R. Gardner (eds.). *Landscape Ecological Analysis*. New York: Springer.

Ahern, J. 2002. *Greenways as Strategic Landscape Planning: Theory and Application.* Ph.D. Thesis. Wageningen University, Wageningen.

Bal, D., H.M. Beije, M. Fellingern, *et al.* 2001. *Handbook Nature Target Types.* LNV Expertise Centre Report Number 2001/020, Wageningen.

Bal, D. and R. Reijnen. 1997. *Nature Policy Practice: Efforts, Effects, Expectations and Chances*. Wageningen: Expertise Centrum LNV.

Beentjes, R.A. and J.C.M. Koopman. 2000. *Pulsing Veins: Giving an Impulse to the Realisation of Ecological Corridors in the Netherlands*. Den Haag: Projectgroup Ecological Corridors.

Bennett, A.F. 1999. *Linkages in the Landscape.* Gland, Switzerland and Cambridge, United Kingdom: The World Conservation Union (IUCN) Forest Conservation Programme.

Bergen Jensen, M., B. Persson, S. Guldager, U. Reeh, and K. Nilsson. 2000. Green structure and sustainability – developing a tool for local planning. *Landscape and Urban Planning* **52**, 117–33.

Broekmeyer, M. and E. Steingrover (eds.). 2001. *Handbook of Robust Corridors and Ecological Prerequisites*. Wageningen: Alterra.

Cabeza, M. and A. Moilanen. 2001. Design of reserve networks and the persistence of biodiversity. *TREE* **16**, 242–8.

Cox, J., R. Kautz, M. MacLaughlin, and T. Gilbert. 1994. *Closing the Gaps in Florida's Wildlife Habitat Conservation System*. Tallahassee: Florida Game and Freshwater Fish Commission.

Etienne, R.S., C.J.F. Ter Braak, and C.C. Vos. 2004. Application of stochastic patch occupancy models to real metapopulations. Pages 105–32 in I. Hanski and O.E. Gaggiiotti (eds.). *Ecology, Genetics, and Evolution of Metapopulations*. San Diego: Elsevier Academic Press.

Golly, F.B. and J. Bellot. 1991. Interactions of landscape ecology, planning and design. *Landscape and Urban Planning* **21**, 3–11.

Haines-Young, R. 2000. Sustainable development and sustainable landscapes: defining a new paradigm for landscape ecology. *Fennia* **178**, 7–14.

Hanski, I. 1999. Habitat connectivity, habitat continuity, and metapopulations in dynamic landscapes. *Oikos* **87**, 209–19.

Henkens, R.J.H.G. 1998. *Ecological Capacity of Ecosystem Types I: The Effect of Outdoor Recreation on Breeding Birds*. IBN-report 363. Wageningen: IBN.

Hill, D., D. Hockin, D. Price, *et al.* 1997. Bird disturbance: improving the quality and utility of disturbance research. *Journal of Applied Ecology* **43**, 275–88.

Hobbs, R.J. 2002. Habitat networks and biological conservation. Pages 150–70 in K.J. Gutzwiller (ed.). *Applying Landscape Ecology in Biological Conservation*. New York: Springer Verlag.

IUCN. 1992. *The Rio Declaration on the Environment*. Gland: IUCN, UNEP, WWF.

Jongman, R.H.G., M. Külvik, and I. Kristiansen. 2003. European ecological networks and greenways. *Landscape and Urban Planning* **68**, 305–19.

Lambeck, R.J. 1999. Landscape planning for biodiversity conservation in agricultural regions. In *Biodiversity Technical Paper Number 2*. Canberra: Environment Australia.

Lambeck, R.J. and R.J. Hobbs. 2002. Landscape and regional planning for conservation. Pages 360–80 in K.J. Gutzwiller (ed.). *Applying Landscape Ecology in Biological Conservation*. New York: Springer.

Linehan, J.R. and M. Gross. 1998. Back to the future, back to basics: the social ecology of landscapes and the future of landscape planning. *Landscape and Urban Planning* **42**, 207–24.

Luck, G.W., G.C. Daily, and P. Ehrlich. 2003. Population diversity and ecosystem services. *TREE* **18**, 331–6.

MANFS. 1990. *Nature Policy Plan 1990*. Den Haag: Ministry of Agriculture, Nature and Food Safety.

MANFS. 2001. *Nature Policy Plan 2001: Nature for People, People for Nature*. Den Haag: Ministry of Agriculture, Nature and Food Safety.

McDonnell, M.-D., H.P. Possingham, I.R. Ball, and E.A. Cousins. 2002. Mathematical methods for spatially cohesive reserve design. *Environmental Modelling and Assessment* **7**, 107–14.

Miller, S.G., R.L. Knight, and C.K. Miller. 1998. Influence of recreational trials on breeding birds communities. *Ecological Application* **8**, 162–9.

Moss, M. 2000. Interdisciplinarity, landscape ecology and the "Transformation of Agricultural Landscapes". *Landscape Ecology* **15**, 303–11.

Nature Policy Agency. 1999. *Nature Balance 1999*. Samsom H.D. Tjeenk Willink, Alphen aan den Rijn, RIVM.

Noss, R.F. and A. Cooperrider. 1994. *Saving Nature's Legacy: Protecting and Restoring Biodiversity*. Washington, DC: Defenders of Wildlife and Island Press.

Opdam, P. 2002. Assessing the conservation potential of habitat networks. Pages 381–404 in K.J. Gutzwiller (ed.). *Applying Landscape Ecology in Biological Conservation*. New York: Springer-Verlag.

Opdam, P., F. Foppen, and C.C. Vos. 2002. Bridging the gap between empirical knowledge and spatial planning in landscape ecology. *Landscape Ecology* **16**, 767–79.

Opdam, P., R. Foppen, R. Reijnen, and A. Schotman. 1995. The landscape ecological approach in bird conservation, integrating the metapopulation concept into spatial planning. *Ibis* **137**, 139–46.

Opdam, P., E. Steingröver, and S. van Rooij. 2005. Ecological networks: a spatial concept for multi-actor planning of sustainable landscapes. *Landscape and Urban Planning* (in press).

Opdam, P., J. Verboom, and R. Pouwels. 2003. Landscape cohesion: an index for the conservation potential of landscapes for biodiversity. *Landscape Ecology* **18**, 113–26.

Opdam, P. and D. Wascher. 2004. Climate change meets habitat fragmentation: linking landscape and biogeographical scale level in research and conservation. *Biological Conservation* **117**, 285–97.

Ovaskainen, O. and I. Hanski. 2004. Metapopulation dynamics in highly fragmented landscapes. Pages 73–104 in I. Hanski and O.E. Gaggiiotti (eds.). *Ecology, Genetics, and Evolution of Metapopulations*. San Diego: Elsevier Academic Press.

Pelk, M., B. Heijkers, R. van Ettiger, *et al.* 1999. *Quality by Connectivity: Why, Where and How*. Wageningen: Ministry of Agriculture, Nature and Food Safety.

Pouwels, R., M.J.S.M. Reijnen, J.T.R. Kalkhoven, and J. Dirksen. 2002. *Ecoprofiles for Species Analysis of Spatial Cohesion with LARCH*. Wageningen: Alterra. Alterra-Report 493.

Ray, N., A. Lehmann, and P. Joly. 2002. Modelling spatial distribution of amphibian populations: a GIS approach based on habitat matrix permeability. *Biodiversity and Conservation* **11**, 2143–65.

Reijnen, R.E., E. van der Grift, M. van der Veen, *et al.* 2000. *The Road to Least Resistance: Priority List of to Be Removed Barriers*. Wageningen: Alterra and Expertise Centrum LNV.

Ricketts, T.H. 2001. The matrix matters: effective isolation in fragmented landscapes. *American Naturalist* **158**, 87–99.

Rothley, K. 2002. Dynamically based criteria for the identification of optimal bioreserve networks. *Environmental Modelling and Assessment* **7**, 123–8.

Steiner, F. 2000. *The Living Landscape: An Ecological Approach to Landscape Planning*. New York: McGraw Hill.

Tewksbury, J.J., D.J. Levey, N.M. Haddad, *et al.* 2002. Corridors affect plants, animals, and their interactions in fragmented landscapes. *PNAS* **99**, 12923–6.

Theobald, D.M., N.T. Hobbs, T. Bearly, *et al.* 2000. Incorporating biological information in local land-use decision-making: designing a system for conservation planning. *Landscape Ecology* **5**, 35–45.

Tress, B., G. Tress, A. Van der Valk, and G. Fry. 2003. *Interdisciplinary and Transdisciplinary Landscape Studies: Potentials and Limitations.* Wageningen: Delta Series 2.

Verboom, J., R. Foppen, P. Chardon, P. Opdam, and P. Luttikhuizen. 2001. Introducing the key-patch approach for habitat networks with persistent populations: an example for marshland birds. *Biological Conservation* **100**, 89–101.

Vos, C.C., H. Baceco, and C.J. Grashof-Bokdam. 2002. Corridors and species dispersal. Pages 84–104 in K.J. Gutzwiller (ed.). *Applying Landscape Ecology in Biological Conservation*. New York: Springer.

Vos, C.C., P. Opdam, and R. Pouwels. 2003. Recreation and biodiversity: a landscape approach. *Landschap* **20**, 3–14.

Vos, C.C., J. Verboom, P.F.M. Opdam, and C.J.F. Ter Braak. 2001. Towards ecologically scaled landscape indices. *American Naturalist* **157**, 24–51.

WCED – World Commission on Environment and Development. 1987. *Our Common Future*. Oxford: Oxford University Press.

Wu, J. and R. Hobbs. 2002. Key issues and research priorities in landscape ecology: an idiosyncratic synthesis. *Landscape Ecology* **17**, 355–65.

Yalden, P.E. and D.W. Yalden. 1990. Recreational disturbance of breeding golden plovers (*Pluvialis apricarius*). *Biological Conservation* **51**, 243–62.

GARY FRY, BÄRBEL TRESS, AND GUNTHER TRESS

14

Integrative landscape research: facts and challenges

14.1 Introduction

There are many tensions in landscape management at spatial scales from individual fields to regions and upwards to global environmental change (Dalgaard *et al.* 2003). Farmers are under increasing pressure to produce non-food products including recreational opportunities, attractive landscapes, and habitats for wildlife. The many different forms of agri-environmental payment schemes are witness to these pressures. In urban landscapes we see a new emphasis on urban green space, urban green structures, and greenways fulfilling multiple goals (Fábos 2004, Gobster and Westphal 2004).

One of the trends in the funding of landscape research over the last 20 years has been the rapid growth of large-scale integrative projects (Höll and Nilsson 1999, Tress *et al.* 2005a). This trend must be seen against the background of environmental concerns that have placed greater demands on the way landscapes are managed and the widening range of objectives they should fulfil. This has fuelled the demand for new research tools to address these problems. Since the problems are complex and span several disciplines, it was natural to consider integrative forms of research as the way forward (Balsiger 2004). In this chapter, we explore several of the major concepts associated with integrative research modes, what funding bodies and researchers expect from such research, and what is being delivered. We discuss the organisational barriers to integration, merit system, and ways to improve the theory base. Finally, we present education and training needs for integrative research and recommend measures to enhance integrative landscape research.

Key Topics in Landscape Ecology, ed. J. Wu and R. Hobbs.

14.2 Methods

This chapter is based on results of the INTELS study investigating Interdisciplinarity and Transdisciplinarity in European Landscape Studies (http://www.intels.cc). To provide a framework for discussing integrative projects and their products, we present data gathered from 19 qualitative interviews with funding bodies, project leaders, and participants involved in integrative projects on European landscapes, as well as contact with 156 journal editors and results from an international web-based survey of 150 researchers involved in integrative landscape research projects. All figures and tables in this chapter are based on the results of these investigations. Additionally, we gathered information from a literature review, reports, and descriptions of research programs. We reviewed the literature on interdisciplinarity and transdisciplinarity, especially theoretical and methodological papers, and those on the practical application of integrative approaches. We collected written material from large national research programs within Europe. The projects were screened to collect statements about the expectation of funding bodies towards integrative projects and, in general, about their understanding of the approaches. We have used this information to review the challenges arising from the rapidly changing field of integrative research with a focus on what we can realistically expect it to achieve. The overall aim of the project is the development of a code of good practice for integrative landscape research (Tress *et al.* 2003a, Tress *et al.* 2005a).

14.3 Defining integrative research approaches

Integrative research approaches, especially interdisciplinarity and transdisciplinarity, are widely used in landscape ecological research. This is true for many other fields of research related to resource management, especially at larger scales, e.g., from landscapes to regions. At these scales, there is a tendency to move away from specialist research disciplines and put a greater emphasis on integrating several, often conflicting, interests, values, and goals. Within landscape research there has been an increasing interest in integrative research approaches (Naveh and Lieberman 1994, Nassauer 1995, Zonneveld 1995, Hobbs 1997, Brandt 2000, Décamps 2000, Klijn and Vos 2000, Palang *et al.* 2000, Naveh 2001, Tress *et al.* 2001, Bastian 2002, Wu and Hobbs 2002).

A major driving force behind the increasing number of integrative research projects has been the emphasis given to integrative research in national and international research funding programs (Tress *et al.* 2005a). Nevertheless, there is much confusion regarding the terminology describing integrative research approaches (Tress *et al.* 2005b). Various terms are used to express interaction between disciplines in landscape research. Expressions include terms such as integrated, holistic, interactive, transepistemic, collaborative, cross-disciplinary or supradisciplinary. Yet most authors using these expressions do not clearly define the meaning of these research approaches. This not only complicates communication of the core concepts, such as interdisciplinarity, but can also make it difficult to match funding body expectations with research achievements. It is paramount to the success of integrative research that agreement over the scope and aims of the project are reached very early on in the project. We stress the need to clarify concepts in the field of integrative research and make definitions explicit. Therefore, we start this chapter with descriptions of how the main concepts are used in this paper; not as an attempt to provide an authoritative set of definitions but as an aid to communication and further the debate on integrative approaches.

- *Disciplinarity*: projects that take place within the bounds of a single, currently recognized academic discipline. We fully appreciate the artificial nature of subject boundaries and that they are dynamic. The project has a disciplinary goal-setting and aims at development of new disciplinary theory and knowledge.
- *Multidisciplinarity*: projects that involve several different academic disciplines researching one theme or problem, but with multiple disciplinary goals. Participants exchange knowledge, but do not aim to cross subject boundaries to create new knowledge and theory. The research process progresses as parallel disciplinary efforts without integration but usually with the aim to compare results. Theory development is discipline oriented.
- *Interdisciplinarity*: projects that involve several unrelated academic disciplines in a way that forces them to cross subject boundaries to create new integrative knowledge and theory and solve a common research goal. By unrelated, we mean that they have contrasting research paradigms. We might consider the differences between qualitative and quantitative approaches or between analytical and interpretative approaches that bring together disciplines from the humanities and the natural sciences.
- *Transdisciplinarity*: projects that both integrate academic researchers from different unrelated disciplines and nonacademic participants, such as land managers and the public, to research a common goal. Transdisciplinary projects create new integrative theory and knowledge among science and society. Transdisciplinarity combines interdisciplinarity with a participatory approach.

In Fig. 14.1, we summarize the key characteristics of nonintegrative (disciplinary and multidisciplinary) and integrative (interdisciplinary

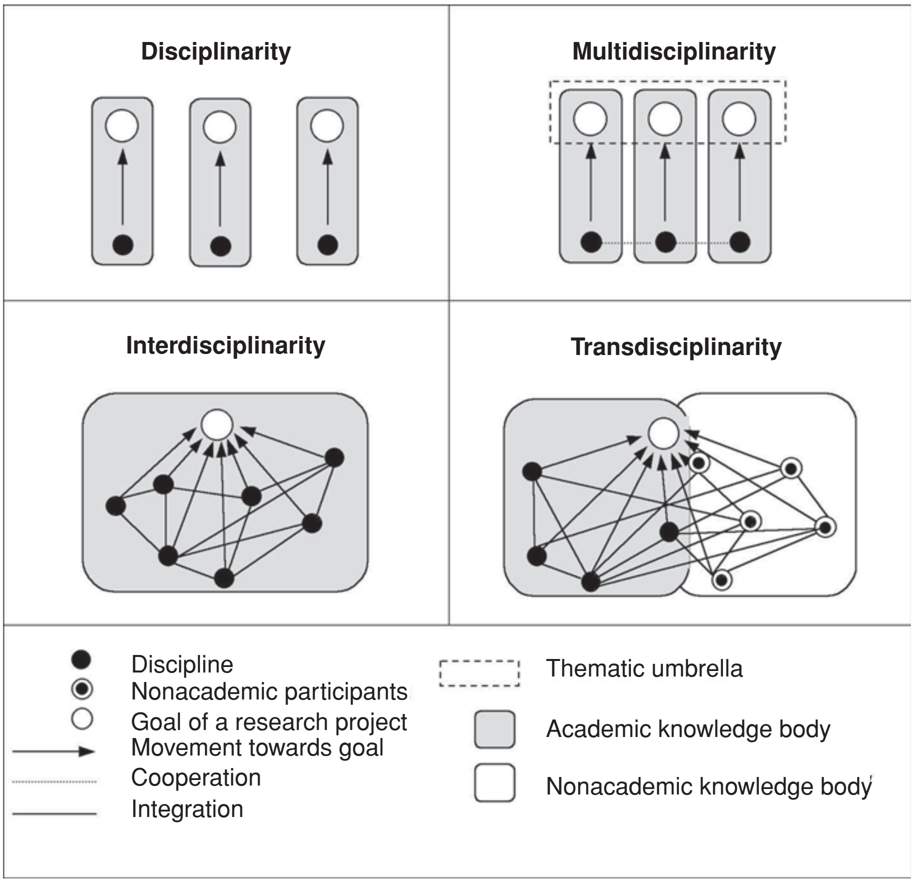


FIGURE 14.1
Main research modes referred to in this chapter showing the differences in degree of integration

and transdisciplinary) research concepts. We further define the concepts of integrative and participatory studies, because both will be necessary to understand the characteristics of the presented research concepts and both will be further used in this chapter. In Fig. 14.2, we visualize the different degrees of integration and stakeholder involvement of integrative and nonintegrative approaches. A detailed overview on all discussed concepts can be found in Tress *et al.* (2005b).

- *Integrative studies*: projects that either work in an interdisciplinary or a transdisciplinary way, in that new knowledge and theory emerges from the *integration* of disciplinary knowledge.
- *Participatory studies*: projects that involve academic researchers and nonacademic participants working together to solve a problem. Academic researchers and nonacademic participants exchange knowledge,

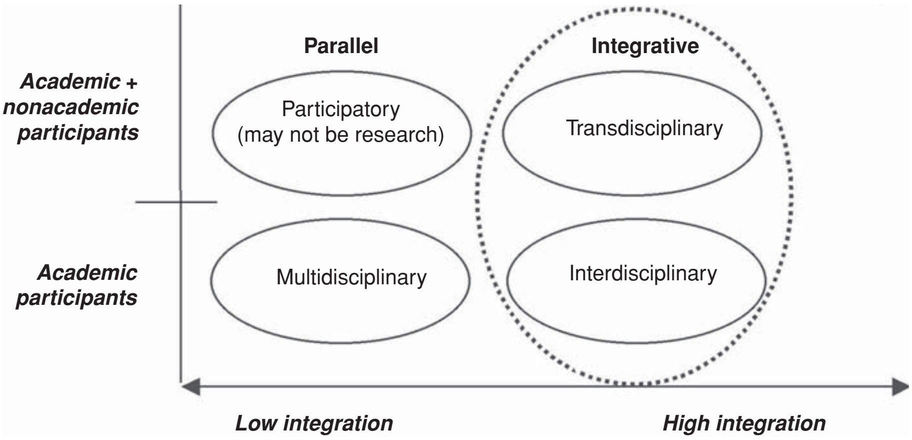


FIGURE 14.2
Degrees of integration and stakeholder involvement in integrative and nonintegrative approaches.

but the focus is not on the integration of the different knowledge cultures to create new knowledge. Both disciplinary and multidisciplinary studies may include nonacademic participants. Participatory studies and especially the use of local knowledge may not necessarily be research, but have an important role in creating engagement and empowerment in the application of scientific findings. It is also under the umbrella of participatory studies that we include the application of scientific results to formulate codes of good practice and other guidelines.

14.4 Motivations for integrative landscape studies

The motivation for participating in integrative studies has frequently arisen from other than academic needs. Individual scientists, project leaders, and research institutes claim that participation in integrative studies is often a response to the priorities of policy-makers and funding bodies. Researchers also claim that their project applications must be integrative to have any chance of winning large research grants in the field of natural resource management. Funding bodies claim that current societal and environmental problems cross policy sectors and disciplinary boundaries and thus call for a common effort. Besides problem-solving, increasing the interaction between science and society as well as building expertise are the key motivations for policy to promote integrative research (Tress *et al.* 2005a; Table 14.1). National research funds are the main driver of the interest in integrative landscape research. This source of funding finances the majority of large-scale integrative

TABLE 14.1. *Expectations of policy and funding bodies*

Expectations
(1) Solving environmental problems • research should provide and apply knowledge to solve pressing environmental problems
(2) Increase the social relevance of research • research should involve stakeholders in problem definition and solution • greater amount of research funding to be used at local level
(3) Scientific progress and expertise • long-term investment in high-quality research • investment in intellectual capital/competitive ability

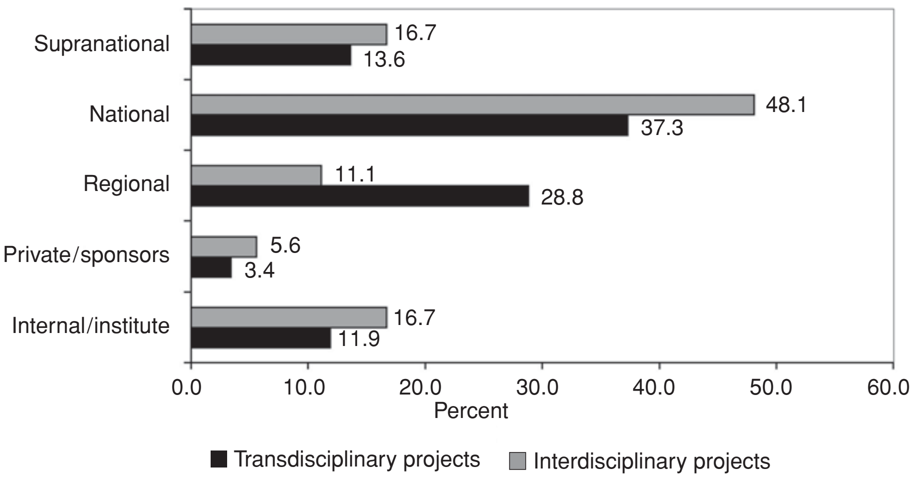


FIGURE 14.3
Sources of funding for interdisciplinary and transdisciplinary projects (data derived from web-based survey of 150 researchers)

studies (see Fig. 14.3). Only for transdisciplinary projects do regional research funds provide a significant proportion of project funding. Transdisciplinary projects frequently engage local or regional stakeholders, which explains the strong involvement of regional funding agencies.

Many of the problems facing landscape management are complex and may involve aspects of animal husbandry, economy, soil science, rural sociology, ecology, and cultural studies, etc. Researchers are expected to contribute to problem solving through examining natural resource management issues from several perspectives. On the other hand, research institutes claim this has made it difficult to fund pure research, even when this is to advance knowledge

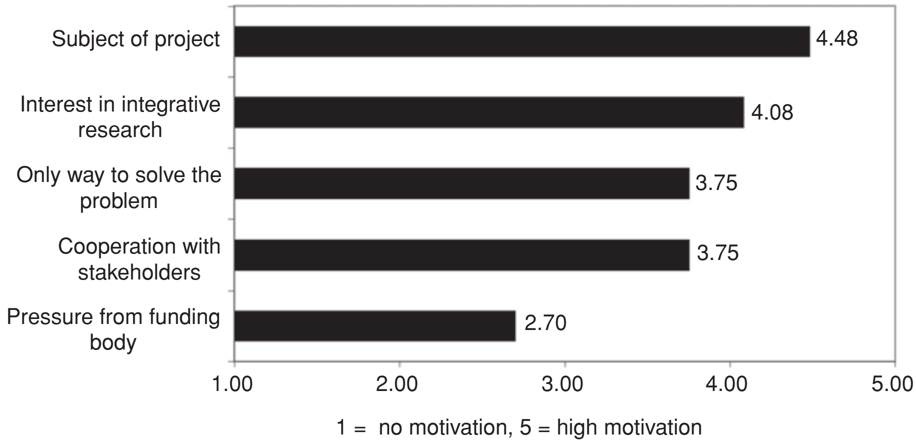


FIGURE 14.4
Top five motivating factors for researchers joining an integrative landscape study (data derived from web-based survey of 150 researchers)

in fields essential to our understanding of sustainable land use. Disciplinary research approaches are considered less able to meet current policy needs, whereas integrative studies are expected to produce a greater proportion of operational solutions (Gibbons *et al.* 1994, Ewel 2001). Researchers have little choice but to follow research policies, and these currently have high expectations of integrative studies. But how realistic are these expectations? The results of our surveys suggest that the expectations are unrealistically high, placing considerable pressure on researchers and their institutes (Tress *et al.* 2003a, 2005a). The key motivation for researchers participating in integrative research is an interest in the subject of the project. Researcher interest in the integration process or in stakeholder participation are lesser motivating factors (Fig. 14.4). Researchers claim that pressure from funding bodies has only a low effect on participation in integrative research. However, interviews with representatives from funding bodies revealed a different picture. Funding bodies are aware that they have a powerful steering role in integrative research and believe that interest would be lower if funds were not used purposely to stimulate integration.

However, the expectations of funding bodies place researchers in a difficult situation. The academic integration of disciplines is both very difficult and can take longer than a single disciplinary research project or program. We require, therefore, a more realistic understanding concerning the nature of integrative research and especially of the limitations of any form of research to solve natural resource management problems (van Asselt and Rijkens-Klomp 2002).

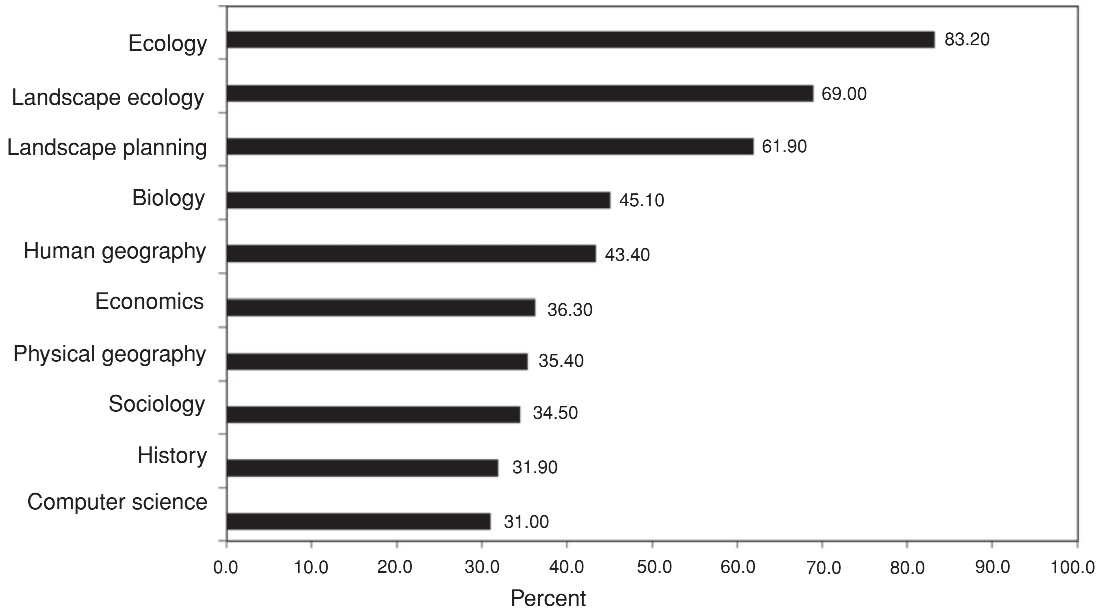


FIGURE 14.5
Top ten disciplines involved in integrative landscape research projects (data derived from web-based survey of 150 researchers)

14.5 What are we trying to integrate?

When conducting integrative research, it is necessary to understand the meaning of disciplines and their boundaries (Klein 1990, Lattuca 2001, Winder 2003) and to critically reflect on their current state and direction (Klein 2004). Disciplines are not static and are increasingly evolving into subdisciplines with their own language and identity. New disciplines appear and old ones disappear, reflecting developments in knowledge cultures and academic institutions. From an epistemological perspective, some boundaries are harder to cross than others (Tress *et al.* 2005b). Integrating humanities and natural science perspectives is especially challenging. For many agricultural research projects, combining economic and ecological perspectives has been difficult, with areas of disagreement related to underlying model assumptions, timescales and what should and should not be taken into account. When undertaking integrative landscape research, such boundaries have to be identified and their nature understood before significant degrees of integration are possible. Disciplines that are most frequently involved in integrative landscape research include ecology, landscape ecology, landscape planning, biology, and human geography (Fig. 14.5). Many of these disciplines were considered as subdisciplines or umbrella disciplines before, but are now independent disciplines represented at university departments and were as such included into our survey.

Integration requires special efforts to bridge academic disciplines and create new knowledge. Most often what we achieve with large projects that span several disciplines and institutes is multidisciplinary research. We are convinced that for many research purposes and for meeting the demands of funding bodies, it can often be the most appropriate research mode. Funding bodies have informed us that they see the process of forcing researchers from different fields to communicate with each other as the main goal of large-scale projects, not necessarily the more difficult task of integrating disciplinary knowledge. They believe that steering researchers to work together, in the same study area or through studying the same problem, will result in formal and informal interactions that will make valuable contributions to solving land-use management problems. Yet, any attempt at achieving a higher degree of integration of disciplines is considered as an add-on benefit that might lead to a new way of solving existing problems.

14.6 Organizational barriers to integration

Achieving a high level of integration between disciplines is difficult, and most projects struggle to realize their intended aim. Project organization, project design, and the day-to-day working environment of people working in large-scale projects can determine success or failure (Jakobsen *et al.* 2004). If institutional frameworks are unsupportive of the integration process as expressed through low resource allocation or cultural isolation, the barriers will be insurmountable. Coordinating the staff of large research teams in space and time is a major challenge of project organization. Spatial separation and infrequent or formal meetings will not help the process of integration. Supportive leadership and management styles combined with frequent and goal-oriented meetings are important factors. Projects that have no clear strategy on how to deal with integration issues often have difficulties in getting started or reaching successful outcomes. We have often observed that projects that set out to be integrative have neither integration goals nor a common problem definition.

The integration process may take longer, especially in defining common research goals, and thus needs more support in the early project phase. Participants should, as far as possible, have the opportunity for regular contact and spontaneous discussion to build the mutual trust and understanding needed to reach high levels of integration. To achieve this, it might be necessary to create temporary environments that physically bring interdisciplinary teams together across institutional boundaries. Research management can do much

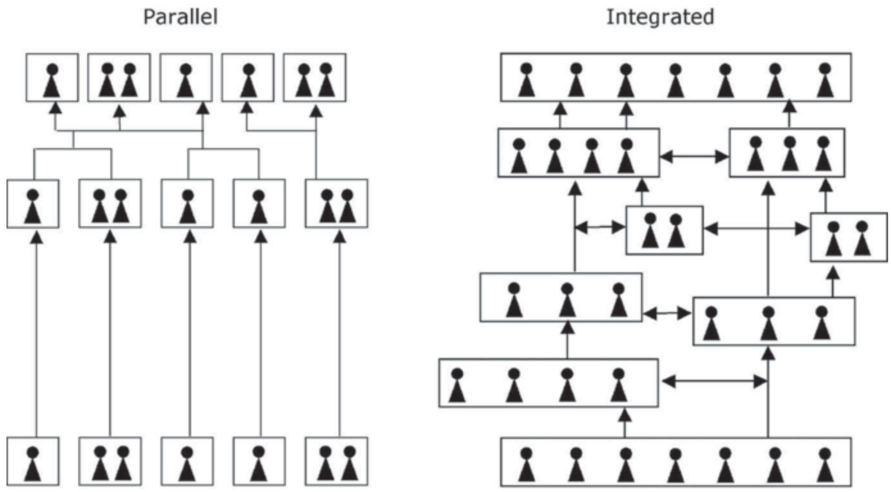


FIGURE 14.6
Fundamental differences in project organization between parallel and integrated research design

to foster inter- and transdisciplinary studies. But this requires research managers to be sensitive to the needs of integrative research and how to create the special conditions and supportive environments needed by integrative research teams. We cannot over-emphasize the role of the project leader. Strong and committed leadership is essential in any new venture. The challenges of integrative research will call for leaders with highly developed interpersonal skills, research credibility, and the ability to maintain the motivation of the team, even when things go wrong.

As a result, many projects end as constellations of small independent (disciplinary) groups. During the course of a project, these groups work more or less independently of each other, only coming together at the end to integrate results. But this is seldom realized when it comes so late in the project process (Fry 2001). The design of integrative research should include measures that span the whole project period. The integration process should start at the beginning – at the stage of project formulation and application. At this stage we need to ask why we are integrating interests and what integration is expected to achieve. If we are unable to formulate answers to these questions, how are we going to know when we have been successful? If the integration does not start at the beginning of a project, this might result in a parallel instead of an integrated project design (Tress *et al.* 2005a; Fig. 14.6). The parallel project design allows for disciplines to carry out their own research agenda and come together at the end of the project to compare results, whereas the integrated design forces teams of mixed researchers to define and agree, at the start of the project, the research agenda needed to address a common research question.

TABLE 14.2. *Training needs for research management at different levels*

Target group/level	Training goal
Research directors/managers	• creating supportive environments • clarifying research strategies/goals • linking strategies to merit system • providing incentives to integrative research
Project leaders	• ensuring budget and time are adequate • facilitating formal and informal meetings • organizing seminars on methods • formulating integration implementation plan
Researchers	• integrating method and theory • organizing integration in daily project work • coping with differences in knowledge cultures • trust building and communication
Ph.D. students	• bridging the demands of a disciplinary Ph.D. and integrative work on a project

14.7 Education and training needs

All research requires a working knowledge of the accepted tools and methods that are integral to specific disciplines and knowledge cultures. Yet, we still observe that interdisciplinary and transdisciplinary studies often start without either participants or project leaders having a firm understanding of integrative research approaches. With no background training or experience in these approaches, researchers often have enormous problems in making the integration work and may return to the relative security of their disciplinary modes of research. To increase the success of integrative research, training is required on different levels and with different goal-settings as is shown in Table 14.2.

The special situation of Ph.D. students in integrative research projects demands special mention. It is common for large-scale integrative projects to involve several Ph.D. students. These students are often given responsibility for the task of achieving integration between the disciplines. Our surveys show that Ph.D. students in integrative projects take longer than average to complete their studies. This may be an especially acute problem for research students in transdisciplinary projects where the solving of a specific practical problem may not involve sufficient research activity or originality to qualify for a Ph.D. Training research students in the epistemological background to integrative research and in the social and psychological processes involved in working across subject and knowledge culture boundaries should be part of their formal course work. It is very important for students to understand the dominant theoretical approaches of the different disciplines involved in a project if they are to play a major role in the integration process (Klein 2004). Our belief is that integrative research is less suited to Ph.D. students and that if they are to tackle this work they will need significantly greater levels of support than is current practice.

14.8 Improving the theory base

Despite high expectations for solving practical land-use problems, interdisciplinarity and transdisciplinarity are just alternative research approaches. This implies that they have an underlying epistemological support, including integrative theories and concepts. However, these are only poorly developed in integrative landscape studies. Wu and Hobbs (2002), Moss (2000) and Antrop (2001) have all pointed out that interdisciplinary work requires method development, conceptual frameworks, and interdisciplinary theory. So far, little coherent interdisciplinary or transdisciplinary theory has emerged from landscape research (Tress and Tress 2002). The same is true for the development of inter- and transdisciplinary concepts and methodologies. One suggestion is to increase efforts in support of a systematic collection of results and experiences of integrative studies in order to identify new generalizable knowledge and to improve methods (Smoliner *et al.* 2001, Fry 2003). The implicit knowledge gained from practical experiences in integrative studies is only rarely made explicit, and is, therefore, unavailable to the scientific community (Nonaka and Takeuchi 1995). This violates a basic academic tradition: to build on existing knowledge. Instead, most integrative landscape studies suffer similar starting problems, especially lack of common methods, and hence progress is slow (Tress *et al.* 2003b).

There are several themes within landscape research that appear especially interesting as starting points for integrative approaches (see Table 14.3). Although different disciplines use their own jargon to describe these concepts, they overlap to a great extent. Exploring the overlapping conceptual zones offers a rich source for the development of common theory.

14.9 The merit system and the products of integrative research

A merit system gives scientists rewards for certain activities that institutes, universities, or the wider scientific community regard as important achievements. Current academic merit systems are tailored for disciplinary approaches and rely heavily on peer-reviewed publications in international journals as the main criteria of success. Likewise, the career advancement of scientists is still mainly based on disciplinary efforts. This is seen by some as a

TABLE 14.3. *Examples of concepts and themes that seem to have wide application to biodiversity, aesthetics and cultural aspects of landscapes (adapted from Fry 2003)*

Concept/theme	Description/application
Connectivity	Connectivity is one of the fundamental processes in landscape ecology. It relates to the functional linkages in a landscape and differs from connectedness, which refers to the physical connections between landscape elements. Connectivity is much more than being physically connected and may include the resistance to movement caused by barriers or by land-use types. Connectivity as a concept is increasingly important in cultural studies where the perception of time and space relate to the mode and characteristics of transport and how these have altered over time. Connectivity in landscapes is an important determinant of the ways in which animals or humans can navigate and move around in the landscape. It will have significance for resource availability and the frequency of cultural interaction and contact with other social groups. The concept involves not just the physical flows across landscapes, but also mental and physical barriers to movement. It also affects visual aspects by indicating accessibility.
Corridors	Corridors are linear landscape features that increase the flow of individuals, materials or energy between resource patches or suitable habitat/settlements and are important in defining the movement infrastructure of both animals and people (Dover 2000). Corridors are one of the most important ways of increasing landscape connectivity. They have been found to function for a wide range of animals and plants (via wind and water spreading of propagules or through seed vectors). The human parallel to landscape corridors is, of course, transport infrastructure. Traffic infrastructure is the most significant human corridor system, increasing access to and availability of physical and mental resources. An important research theme in geography is the way transport infrastructure and modes of transport affect our concepts of place and space.
Nodes	Nodes are intersections in movement corridors that result in important meeting places. They are especially of significance for determining the number of alternative routes individuals can take to move around a landscape. They are also important in visual orientation in built and natural landscapes. Nodes are therefore important in determining flows of species, nutrients, and energy around landscapes. If we examine a typical hedgerow network, we find nodes where hedges meet or cross. It is interesting to note that for a wide range of animal and plant taxa, the number of species that accumulate at nodes is richer than the surrounding landscape (Fry 1991, Sarlöv-Herlin and Fry 2000). The ecological interpretation of this phenomenon is that the diversity of species is a result of the favorable (food, shelter, and less disturbance) habitat at nodes and their significance for landscape-scale dispersal mechanisms. Movement models predict greater visitation rates to nodes than other sections of hedgerow networks. The importance of nodes to geographers and archaeologists is much older than the new-found relations in ecology. The role of nodes in transport infrastructure has long been a major explanatory variable in the locating of defences, settlements, and industry. Multiple infrastructure nodes such as waterways and roads have been especially important at different historical periods.

Supplementation and Complementation	Supplementation and complementation are key landscape ecological concepts that reflect different strategies by which individuals and populations sustain themselves with essential resources in fragmented landscapes (Taylor *et al.* 1993). The process of supplementation relates to obtaining necessary resources from several small sources within accessible distance. We can think of an animal such as the wolf that has a large home range; it may be able to use several forest areas to meet all its needs as long as the forests are all accessible and within traveling distance. Complementation is a similar, but subtly different, concept that describes the acquisition of several quite different but essential resources, which must be "available" i.e., close enough to utilize without expending too much energy or being exposed to risks such as predation. "Availability" of resources in human terms is related to both physical and cultural/ behavioral aspects. It is thus possible to discuss the theory of complementation in archaeological terms as the basis for many locational models (Fry *et al.* 2004).
Heterogeneity	Heterogeneity is a complex concept and much used in landscape analysis for a wide range of purposes. Heterogeneity sums up two aspects of landscapes: their grain size (the size of fields, forest patches etc.), and the variety that exists. Perhaps because of this dual influence on heterogeneity, it is not so easy to capture by numerical indices (Dramstad *et al.* 2001, Fjellstad *et al.* 2001). Despite difficulties in capturing heterogeneity, it ensures that a wide range of resources are available in a spatially restricted area, and is also one of the dominant visual aspects of landscapes and correlated to a wide range of animal and plant population and community characteristics. Because of this, landscape heterogeneity has been a central theme in the development of landscape indicators for biodiversity (Schneider and Fry 2001, Leitão and Ahern 2002, Olff and Richie 2002). It is also a major feature in human landscape preference studies and used in landscape characterization. In a paper based on the results of a multiple-interest landscape monitoring project, Dramstad *et al.* (2001) found that bird and plant diversity, the density of prehistoric grave mounds and human visual preferences all responded negatively to the grain size of the landscapes. In other words, the finer the scale of the landscape the higher the biodiversity, cultural heritage, and visual value of the landscape. The concept of heterogeneity is thus likely to provide an interesting starting point for theory generation across disciplinary boundaries in an attempt to explain its wide influence in humans and nature at the landscape level.

TABLE 14.4. *Statements by researchers related to their experiences of trying to publish the results of integrative landscape research (Tress* et al. 2005; *Data derived from qualitative interviews)*

No.	Statement
1	For interdisciplinary or transdisciplinary studies you cannot find the right journals
2	Publications from interdisciplinary and transdisciplinary projects are not suited for journal publication
3	It becomes more difficult to publish in journals with high-impact factor, the more applied the study is

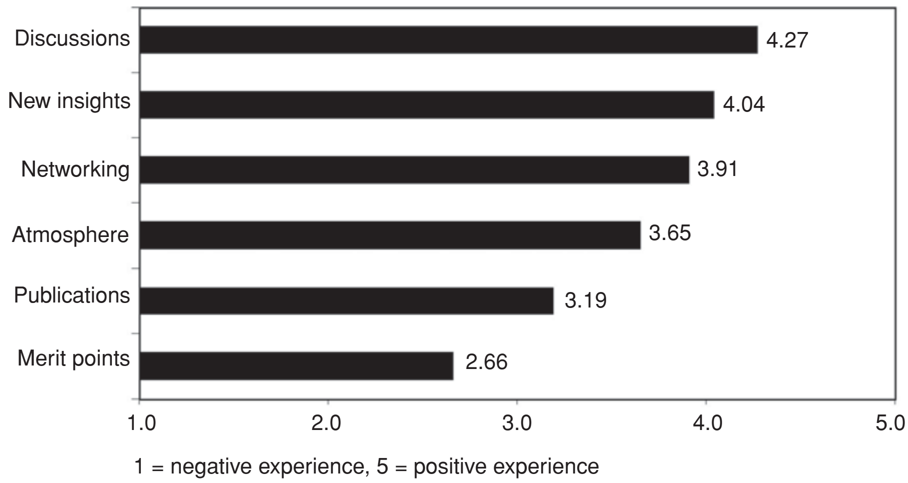


FIGURE 14.7
Some of the positive and negative experiences of researchers working in integrative landscape research (data derived from web-based survey of 150 researchers)

limitation to the development of integrative approaches. If scientists are to work with integrative approaches, their involvement should have an equal chance of being rewarded as disciplinary efforts. In Fig. 14.7, we present what researchers consider as negative and positive experiences of integrative research; merit points and publications are ranked more negatively than other effects. A merit system for integrative approaches may require academia to acknowledge a wider range of research products. Assessment of these products, however, will need the development of extensive, systematic, transparent, and fair systems of peer-reviewed achievement. There exists confusion over what integrative research can deliver. There is a wide belief among researchers that it is difficult to publish the results of integrative research (Table 14.4). To test this, we contacted more than 156 journals in landscapes, agriculture, forestry,

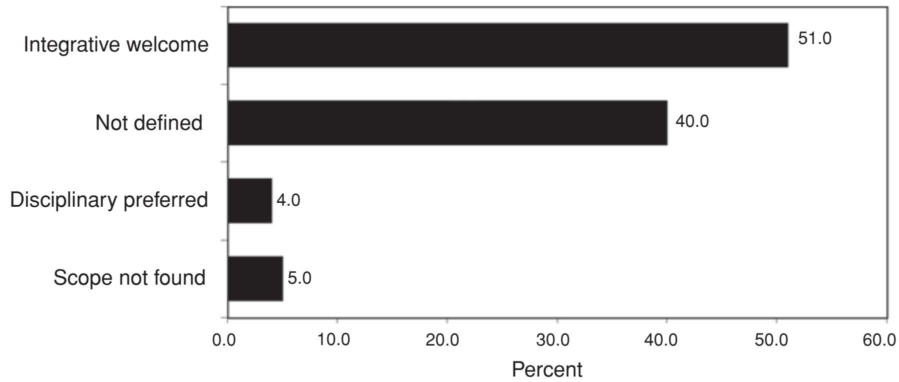


FIGURE 14.8
Editorial policies towards publishing integrative research papers of 156 journals that publish landscape research; as extracted from their published instructions to authors (n = 156)

ecology, planning, and cultural studies. All have published papers from landscape research. We asked the editors of these journals whether they would publish the results of integrative studies. Of the 97 that replied 96 said that they would welcome such papers. Similarly, the instructions to authors of scientific journals show that more than 50 percent actively seek papers from interdisciplinary research and a further 40 percent are neutral (Fig. 14.8). There may be discrepancies between the declared publishing policy of journals and the responses of reviewers to integrative papers, or there may be other reasons for finding the results of integrative research difficult to publish. Researchers in our survey mentioned the factors listed in Table 14.5.

An equally important product of integrative research is its ability to solve environmental problems. However, we have found little evidence to suggest that integrative projects are more or less likely than single-disciplinary research to provide solutions to environmental problems. Applied research is more likely to focus on specific problems and their solutions, but whether integrative approaches result in more or better solutions is difficult to assess. There is little empirical evidence either way. The long-term benefits of increasing communication between disciplines are also difficult to assess.

14.10 Mapping the boundaries of research

At an abstract level, integrative or participatory research projects contain elements of knowledge creation, application, and reflection, as well as feedback to science. These processes go hand in hand and mutually influence each other. We have analyzed the process of knowledge creation in two steps to clarify the boundary between research and consultancy/outreach activities (Tress *et al.* 2003b; Fig. 14.9):

TABLE 14.5. *Statements by researchers concerning why integrative research may not get published (Tress* et al. *2005; Data derived from qualitative interviews)*

No.	Reason
1	The work does not make a significant or novel contribution to the relevant bodies of knowledge
2	The work is too descriptive
3	It is difficult to write joint papers when co-authors belong to different research cultures – might take more time. Different knowledge cultures mean: • Different styles in writing • Different concepts of data • Different strategies for analyzing data • Lack of common theory base
4	The work is the application of existing knowledge, not original enough to be published
5	Large-scale interdisciplinary projects often lack replication or control or may study a single case making it difficult to generalize and be accepted by high-ranking journals

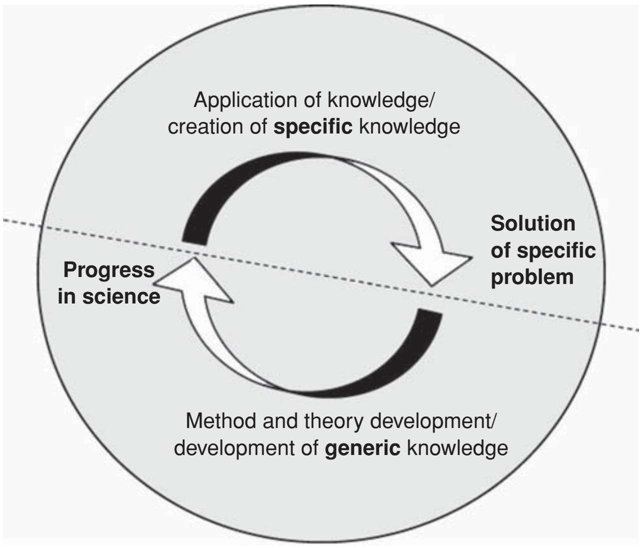


FIGURE 14.9
The process of knowledge creation – from problem solving to progress in science (Tress *et al.* 2003b)

1. Existing knowledge is used to develop a solution to a specific problem. This knowledge may be derived from the collective expertise of the project team (which may include nonacademics as well as academics) or from the results of earlier research studies – part of the body of scientific knowledge. For a project to be considered as research demands that new knowledge has to be generated by the project team in order to solve the problem. This debate is very relevant to the increased consultancy and outreach activities of European research institutes.
2. The second part of the process of knowledge creation occurs when the focus is on the generation of *generic* knowledge. We also acknowledge that the systematic application of existing knowledge can be a form of hypothesis testing – leading to the production of generic knowledge. As science is interested in the nature and behavior of observable phenomena (Feynman 1998), it seeks knowledge that has relevance and validity beyond a specific context. This generic knowledge is fed back to science usually through the publication of a peer-reviewed scientific paper or book and is the main process through which progress in science takes place.

It would appear that many applied integrative projects only focus on the goal of gaining the knowledge needed for solving the specific problem defined by the funding agency. Once this has been achieved, there may be neither time nor money for more basic reflection on the knowledge created or how it relates to the wider scientific context. The focus of consultancy work is more on the application of existing knowledge than on the creation of generic knowledge and hence scientific advancement. Consultancy relies on the application of existing knowledge for the solution of a problem – the work is not usually considered as research even when that solution is contextual and unique. The difference between fundamental research and consultancy is also illustrated in Fig. 14.10. Research projects, in their intention to be applied or solve a specific problem, may transgress the border between research and consultancy.

14.11 Enhancing integrative landscape ecology research

What is a good interdisciplinary or transdisciplinary study? Attempts have been made to develop sets of evaluation criteria for integrative projects (Defila and Di Giulio 1998, Spaapen and Wamelink 1999, Balsiger 2004; see Table 14.6). There are, however, no widely recognized quality standards that could be used to evaluate projects through their various stages. Quality standards would have two main advantages. Firstly, agreed standards would make it easier for funding bodies to distinguish real interdisciplinary projects from

TABLE 14.6. *List of measures to increase the success of integrative research projects based on research of the INTELS project (Tress* et al. *2003a, 2005)*

No.	Measure
1	Start the work early in the process with a plan for integration
2	Give participants opportunity for frequent formal and informal contact
3	Identify shared problems/challenges as work packages
4	Organize seminars specifically to communicate different research modes and approaches and to identify common theory and methods
5	Include a project plan for integrated products showing add-on value
6	In general small groups work better than large
7	The power of personal chemistry cannot be over-estimated
8	Good project leadership and management are essential to the success of large projects
9	Supportive institutional structures are required, these provide reward and identity
10	Training is required at the researcher, project leader, and institutional management level
11	Integrative projects may not be suitable for Ph.D. studies

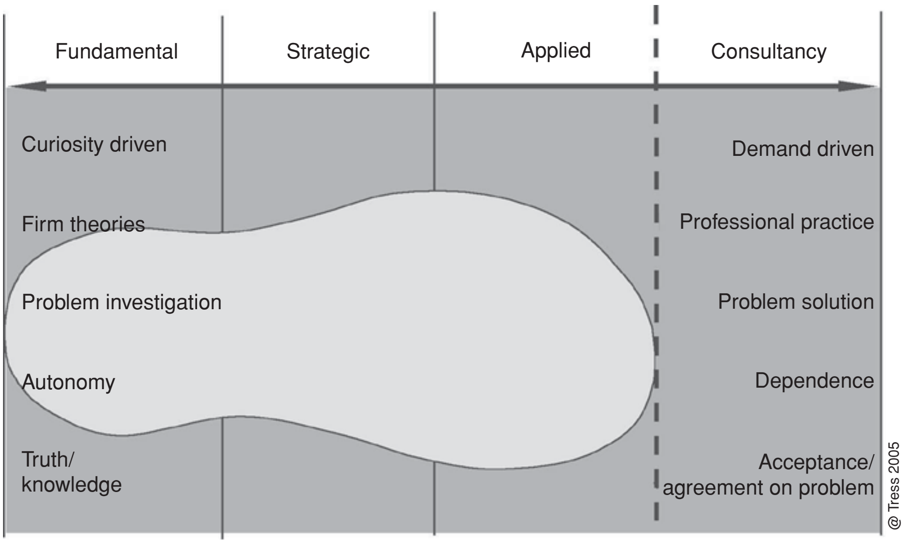


FIGURE 14.10
Defining the footprint of integrative research – from pure science to professional practice

those that only play with the name to improve their chance for getting support. Secondly, standards would also serve as guidelines for researchers when setting criteria and goals for project planning. Development of integrative research standards would contribute significantly to improving interdisciplinary and transdisciplinary projects. One of the most important of these criteria would be the degree of integration reached in a project and how this contributed to the add-on value of end products.

Good personal chemistry between researchers is also a key to success. Mutual trust, motivation, and pleasure in working together are important in any research team. However, the common ground is much smaller when contrasting disciplines are involved than when researchers all come from the same or closely related disciplines. Overcoming cultural barriers places high demands on the interpersonal relationships between members of interdisciplinary projects. As a result, smaller research teams are often better suited to crossing disciplinary boundaries than larger ones.

One characteristic of the large-scale research projects studied by the INTELS project was that most had no specific plan to reach integration. Yet, many of the other aims of these projects had clear objectives linked to specific methods and milestones to assess planned progress. We firmly believe that achieving integration should also be seen as a specific project aim with a full description of the planned progress of integration and how this will be achieved. Only in a few recently started initiatives, have we identified specific aims for improving knowledge and skills in integrative research among the aims of research programs.

14.12 Conclusion

Landscape ecology has undoubtedly been a major force in the development of integrative landscape research. This contribution to the development of integrative approaches is widely evidenced by the many landscape conferences that have held special sessions or had integration as a major theme as reported by Brandt (2000), Klijn and Vos (2000), Moss (2000), Tress *et al.* (2001), and Mander *et al.* (2004). The overarching concepts presented in Section 14.8 of this paper are but a few of those emerging within landscape ecology. Developing these concepts may lead to significant advances in the development of integrative landscape theory. We also urgently need new methods for use in integrative research. The ways different disciplines collect and analyze data are both different and often incompatible. Yet, here again we see new research initiatives that combine qualitative and quantitative approaches, and traditional knowledge with landscape metrics. The future will include a rich variety of such studies that will redefine the scope and direction of landscape ecology. We have seen the rapid increase in integrative landscape research projects. Many of these are large in terms of budget and personnel resources. The time is near when we can start to compile a meta-analysis of such projects to locate aspects of emerging theory and identify robust methodological approaches. The science of landscape ecology requires this academic growth as well as its contribution finding practical solutions to resource management issues.

To further improve the success rate of integrative landscape studies, the expectations of scientists and funding bodies need to reach a better balance and be made explicit. This would include greater reflection on and a more realistic appraisal of what integrative research approaches can achieve and what they cannot (see *Futures* **36**, issue (**4**), 2004). We need to acknowledge – against the tide of opinion – that integrative studies are not the solution to each and every land management problem nor will they always result in win–win resource-management situations. Working interdisciplinarily or transdisciplinarily will not prevent power struggles between interest groups and will not tell policy-makers what should be done. Integrative research approaches will, however, increasingly inform policy-makers of the consequences of different land-use scenarios for a wider range of landscape values and stakeholders and hence provide a better basis for decision-making.

In this chapter we have identified many of the challenges and difficulties facing integrative research. Despite the problems, we have also noted that it is possible to find successful interdisciplinary and transdisciplinary projects and that such projects increasingly succeed in publishing their results. In addition, participants from integrative studies report that their involvement gave them unexpected new insights into other fields of research and also into their own subjects (Kinzig 2001, Tress *et al.* 2005a). These insights sometimes fundamentally changed the way researchers perceived their own discipline or their methodological approach. To all those involved in integrative research from planning and policy levels to individual researchers, the future challenge is to further elaborate on the improvement and advancement of integrative research, to bring its full benefit to both academia and society.

References

Antrop, M. 2001. The language of landscape ecologists and planners: a comparative content analysis of concepts used in landscape ecology. *Landscape and Urban Planning* **55**, 163–73.

Balsiger, P. W. 2004. Supradisciplinary research practices: history, objectives and rationale. *Futures* **36**, 407–21.

Bastian, O. 2002. Landscape ecology: towards a unified discipline? *Landscape Ecology* **16**, 757–66.

Brandt, J. 2000. Editorial: the landscape of landscape ecologists. *Landscape Ecology* **15**, 181–5.

Dalgaard, T., N. J. Hutchings, and J. R. Porter. 2003. Agroecology, scaling and interdisciplinarity. *Agriculture Ecosystems and Environment* **100**, 39–51.

Décamps, H. 2000. Demanding more of landscape research (and researchers). *Landscape and Urban Planning* **47**, 105–9.

Defila, R. and A. Di Giulio. 1998. Interdisziplinarität und Disziplinarität. Pages 111–37 in J.H. Olbertz (ed.). *Zwischen den Fächern – über den Dingen?* Opladen: Leske und Budrich.

Dover, J. 2000. Human, environmental and wildlife aspects of corridors with specific reference to UK planning practice. *Landscape Research* **25**, 333–44.

Dramstad, W.E., G. Fry, W.J. Fjellstad, *et al.* 2001. Integrating landscape-based values. *Landscape and Urban Planning* **57**, 257–68.

Ewel, K.C. 2001. Natural resource management: the need for interdisciplinary collaboration. *Ecosystems* **4**, 716–22.

Fábos, J.G. 2004. Greenway planning in the United States: its origins and recent case studies. *Landscape and Urban Planning* **68**, 321–42.

Feynman, R. 1998. *The Meaning of Everything*. London: Penguin Books.

Fjellstad, W.J., W.E. Dramstad, G.-H. Strand, and G.L.A. Fry. 2001. Heterogeneity as a measure of spatial pattern for monitoring agricultural landscapes. *Norwegian Journal of Geography* **55**, 71–6.

Fry, G., B. Skar, G.B. Jerpåsen, V. Bakkestuen, and L. Erilestad. 2004. Predicting archaeological sites: a method based on landscape indicators. *Landscape and Urban Planning* **67**, 97–107.

Fry, G.L.A. 1991. Conservation in agricultural ecosystems. Pages 415–43 in I.F. Spellerberg, F.B. Goldsmith, and M.G. Morris (eds.). *The Scientific Management of Temperate Communities for Nature Conservation*. London: Blackwell.

Fry, G.L.A. 2001. Multifunctional landscapes: towards transdisciplinary research. *Landscape and Urban Planning* **57**, 159–68.

Fry, G.L.A. 2003. From objects to landscapes in natural and cultural heritage management: a role for landscape interfaces. Pages 237–53 in H. Palang and G. Fry (eds.). *Landscape Interfaces: Cultural Heritage in Changing Landscapes*. Dordrecht: Kluwer.

Gibbons, M., C. Limoges, H. Nowotny, *et al.* 1994. *The New Production of Knowledge: The Dynamics of Science and Research in Contemporary Societies*. London: Sage.

Gobster, P.H. and L.M. Westphal. 2004. The human dimensions of urban greenways: planning for recreation and related experiences. *Landscape and Urban Planning* **68**, 147–65.

Hobbs, R. 1997. Future landscapes and the future of landscape ecology. *Landscape and Urban Planning* **37**, 1–9.

Höll, A. and K. Nilsson. 1999. Cultural landscape as subject to national research programmes in Denmark. *Landscape and Urban Planning* **46**, 15–27.

Jakobsen, C.H., T. Hels, and W.J. McLaughlin. 2004. Barriers and facilitators to integration among scientists in transdisciplinary landscape analysis: a cross country comparison. *Forest Policy and Economics* **6**, 15–31.

Kinzig, A.P. 2001. Bridging disciplinary divides to address environmental and intellectual challenges. *Ecosystems* **4**, 709–15.

Klein, J.T. 1990. *Interdisciplinarity: History, Theory and Practice*. Detroit: Wayne State University Press.

Klein, J.T. 2004. Prospects for transdisciplinarity. *Futures* **36**, 515–26.

Klijn, J. and W. Vos, (eds.). 2000. *From Landscape Ecology to Landscape Science*. Dordrecht: Kluwer.

Lattuca, L.R. 2001. *Creating Interdisciplinarity: Interdisciplinary Research and Teaching among College and University Faculty*. Nashville: Vanderbilt University Press.

Leitão A.B. and J. Ahern. 2002. Applying landscape ecological concepts and metrics in sustainable landscape planning. *Landscape and Urban Planning* **59**, 65–93.

Mander, Ü., H. Palang, and M. Ihse. 2004. Editorial: development of European landscapes. *Landscape and Urban Planning* **67**, 1–8.

Moss, M. 2000. Interdisciplinarity, landscape ecology and the "Transformation of Agricultural Landscapes." *Landscape Ecology* **15**, 303–11.

Nassauer, J.I. 1995. Culture and changing landscape structure. *Landscape Ecology* **10**, 229–37.

Naveh, Z. 2001. Ten major premises for a holistic conception of multifunctional landscapes. *Landscape and Urban Planning* **57**, 269–84.
Naveh, Z. and A. Lieberman. 1994. *Landscape Ecology: Theory and Application*. 2nd edn. Berlin, Heidelberg: Springer.
Nonaka, I. and H. Takeuchi. 1995. *The Knowledge-creating Company*. Oxford: Oxford University Press.
Olff, H. and M.E. Richie. 2002. Fragmented nature: consequences for biodiversity. *Landscape and Urban Planning* **58**, 83–92.
Palang, H., Ü. Mander, and Z. Naveh. 2000. Holistic landscape ecology in action. *Landscape and Urban Planning* **50**, 1–6.
Sarlöv-Herlin, I. and G. Fry. 2000. Dispersal of woody plants in forest edges and hedgerows in a Southern Swedish agricultural area: the role of site and landscape structure. *Landscape Ecology* **15**, 229–42.
Schneider, C. and G.L.A. Fry. 2001. The influence of landscape grain size on butterfly diversity in grasslands. *Journal of Insect Conservation* **5**, 163–71.
Smoliner, C., R. Häberli, and M. Welti. 2001. Mainstreaming transdisciplinarity: a research-political campaign. Pages 263–71 in J.T. Klein, W. Grossenbacher-Mansu, R. Häberli, A. Bill, R.W. Scholz, and M. Welti (eds.). *Transdisciplinarity: Joint Problem Solving among Science, Technology, and Society*. Basel: Birkhäuser.
Spaapen, J.B. and F.J.M. Wamelink. 1999. *The Evaluation of University Research: A Method for the Incorporation of Societal Value of Research*. The Hague: NRLO-report 99/12.
Taylor, P.D., L. Fahrig, K. Henein, and G. Merriam. 1993. Connectivity is a vital element of landscape structure. *Oikos* **68**, 571–3.
Tress, B. and G. Tress. 2002. Disciplinary and meta-disciplinary approaches in landscape ecology. Pages 25–37 in O. Bastian and U. Steinhardt (eds.). *Development and Perspectives in Landscape Ecology*. Dordrecht: Kluwer.
Tress, B., G. Tress, H. Décamps, and A. d'Hauteserre. 2001. Bridging human and natural sciences in landscape research. *Landscape and Urban Planning* **57**, 137–41.
Tress, B., G. Tress, and G. Fry. 2005a. Integrative studies on rural landscapes: policy expectations and research practice. *Landscape and Urban Planning* **70**, 177–91.
Tress, G., B. Tress, and G. Fry. 2005b. Clarifying integrative research concepts in landscape ecology. *Landscape Ecology* **20**, 479–93.
Tress, B., G. Tress, A. Van der Valk, and G. Fry (eds.). 2003a. *Interdisciplinarity and Transdisciplinarity in Landscape Studies: Potential and Limitations*. Wageningen: Delta Series 2.
Tress, G., B. Tress, and G. Fry. 2003b. Knowledge creation and reflection in integrative and participatory projects. Pages 14–24 in G. Tress, B. Tress, and M. Bloemmen (eds.). *From Tacit to Explicit Knowledge in Integrative and Participatory Research*. Wageningen: Delta Series 3.
van Asselt, M.B.A. and N. Rijkens-Klomp. 2002. A look in the mirror: reflection on participation in integrated assessment from a methodological perspective. *Global Environmental Change* **12**, 167–84.
Winder, N. 2003. Successes and problems when conducting interdisciplinary or transdisciplinary (= integrative) research. Pages 74–90 in B. Tress, G. Tress, A.V.d. Valk, and G. Fry (eds.). *Interdisciplinarity and Transdisciplinarity in Landscape Studies: Potential and Limitations*. Wageningen: Delta Series 2.
Wu, J. and R. Hobbs. 2002. Key issues and research priorities in landscape ecology: an idiosyncratic synthesis. *Landscape Ecology* **17**, 355–65.
Zonnenveld, I.S. 1995. *Land Ecology: An Introduction to Landscape Ecology as a Base for Land Evaluation, Land Management and Conservation*. Amsterdam: SPB.

PART III

Synthesis

JIANGUO WU AND RICHARD HOBBS

15

Landscape ecology: the state-of-the-science

15.1 Introduction

Good science starts with precise definitions because clearly defined terminology is a prerequisite for any fruitful scientific discourse. For rapidly developing interdisciplinary sciences like landscape ecology, unambiguous definitions are particularly important. Contemporary landscape ecology is characterized by a flux of ideas and perspectives that cut across a number of disciplines in both natural and social sciences, as evidenced in the previous chapters of this volume. On the one hand, after having experienced an unprecedented rapid development in theory and practice in the past two decades, landscape ecology has become a globally recognized scientific enterprise. On the other hand, more than 65 years after the term "landscape ecology" was first introduced, landscape ecologists are still debating on what constitutes a landscape and what landscape ecology really is (e.g., Wiens 1992, Hobbs 1997, Wiens and Moss 1999, Wu and Hobbs 2002).

Two major schools of thought in landscape ecology have widely been recognized: the European approach that is more humanistic and holistic and the North American approach that is more biophysical and analytical. To increase the synergies between the two approaches, not only do we need to appreciate the values of both approaches, but also to develop an appropriate framework in which different perspectives and methods are properly related. Toward this end, in this chapter we shall compare and contrast the European and North American approaches through several exemplary definitions (see Table 15.1). We shall argue that both approaches can be traced back to the original definition of landscape ecology, and that recent developments seem to show a tendency for unification of once diverging perspectives. Then, we shall propose a

Key Topics in Landscape Ecology, ed. J. Wu and R. Hobbs.

TABLE 15.1. *A list of exemplary definitions of landscape ecology*

Definition	Source
The German geographer Carl Troll coined the term "landscape ecology" in 1939, and defined it in 1968 as "the study of the main complex causal relationships between the life communities and their environment in a given section of a landscape. These relationships are expressed regionally in a definite distribution pattern (landscape mosaic, landscape pattern) and in a natural regionalization at various orders of magnitude" (Troll 1968; cited in Troll 1971).	• Troll, C. 1939. Luftbildplan and okologische bodenforschung. *Zeitschraft der Gesellschaft fur Erdkunde Zu Berlin* 241–98. • Troll, C. 1968. Landschaftsokologie. Pages 1–2 in *Pflanzensoziologie und Landschaftsokologie – Symposium Stolzenau*. Junk: The Hague. • Troll, C. 1971. Landscape ecology (geoecology) and biogeocenology – a terminological study. *Geoforum* **8**, 43–6.
"Landscape ecology is an aspect of geographical study which considers the landscape as a holistic entity, made up of different elements, all influencing each other. This means that land is studied as the 'total character of a region' , and not in terms of the separate aspects of its component elements" (Zonneveld 1972).	• Zonneveld, I. S. 1972. *Land Evaluation and Land(scape) Science*. Enschede, The Netherlands: International Institute for Aerial Survey and Earth Sciences.
"Landscape ecology is a young branch of modern ecology that deals with the interrelationship between man and his open and built-up landscapes" based on general systems theory, biocybernetics, and ecosystemology (Naveh and Liberman 1984). "Landscapes can be recognized as tangible and heterogeneous but closely interwoven natural and cultural entities of our total living space," and landscape ecology is "a holistic and transdisciplinary science of landscape study, appraisal, history, planning and management, conservation, and restoration" (Naveh and Liberman 1994).	• Naveh, Z. and A. S. Lieberman. 1984. *Landscape Ecology: Theory and Application*. New York: Springer-Verlag. • Naveh, Z. and A. S. Lieberman. 1994. *Landscape Ecology: Theory and Application*, 2nd edn. New York: Springer-Verlag.
"A landscape is a kilometers-wide area where a cluster of interacting stands or ecosystems is repeated in similar form; landscape ecology, thus, studies the structure, function and development of landscapes" (Forman 1981). Landscape structure refers to "the spatial relationships among the distinctive ecosystems;" landscape function refers to "the flows of energy, materials, and species among the component ecosystems;" and landscape change refers to "the alteration in the structure and function of the ecological mosaic over time" (Forman and Godron 1986).	• Forman, R. T. T. 1981. Interaction among landscape elements: a core of landscape ecology. Pages 35–48 in S. P. Tjallingii and A. A. de Veer (eds.). *Perspectives in Landscape Ecology*. Wageningen: Pudoc. • Forman, R. T. T. and M. Godron. 1986. *Landscape Ecology*. New York: John Wiley & Sons Inc.

"Landscape ecology focuses explicitly upon spatial pattern. Specifically, landscape ecology considers the development and dynamics of spatial heterogeneity, spatial and temporal interactions and exchanges across heterogeneous landscapes, influences of spatial heterogeneity on biotic and abiotic processes, and management of spatial heterogeneity" (Risser *et al.* 1984). "Landscape ecology is not a distinct discipline or simply a branch of ecology, but rather is the synthetic intersection of many related disciplines that focus on the spatial-temporal pattern of the landscape" (Risser *et al.* 1984).	• Risser, P. G., J. R. Karr, and R. T. T. Forman. 1984. *Landscape Ecology: Directions and Approaches*. Special Publication 2. Champaign: Illinois Natural History Survey.
"Landscape ecology emphasizes broad spatial scales and the ecological effects of the spatial patterning of ecosystems" (Turner 1989).	• Turner, M. G. 1989. Landscape ecology: the effect of pattern on process. *Annual Review of Ecology and Systematics* **20**, 171–97.
"Landscape ecology is the study of the reciprocal effects of the spatial pattern on ecological processes," and "concerns spatial dynamics (including fluxes of organisms, materials, and energy) and the ways in which fluxes are controlled within heterogeneous matrices" (Pickett and Cadenasso 1995).	• Pickett, S. T. A. and M. L. Cadenasso. 1995. Landscape ecology: spatial heterogeneity in ecological systems. *Science* **269**, 331–4.
"Landscape ecology investigates landscape structure and ecological function at a scale that encompasses the ordinary elements of human landscape experience: yards, forests, fields, streams, and streets" (Nassauer 1997).	• Nassauer, J. I. 1997. Culture and landscape ecology: insights for action. Pages 1–11 in J. I. Nassauer (ed.). *Placing Nature: Culture and Landscape Ecology*. Washington, DC: Island Press.
Landscape ecology is "ecology that is spatially explicit or locational; it is the study of the structure and dynamics of spatial mosaics and their ecological causes and consequences" and "may apply to any level of an organizational hierarchy, or at any of a great many scales of resolution" (Wiens 1999).	• Wiens, J. A. 1999. Toward a unified landscape ecology. Pages 148–51 in J. A. Wiens and M. R. Moss (eds.). *Issues in Landscape Ecology*. Snowmass Village: International Association for Landscape Ecology.
"Landscape ecology emphasizes the interaction between spatial pattern and ecological process, that is, the causes and consequences of spatial heterogeneity across a range of scales" (Turner *et al.* 2001). "Two important aspects of landscape ecology . . . distinguish it from other subdisciplines within ecology": "First, landscape ecology explicitly addresses the importance of spatial configuration for ecological processes" and "second, landscape ecology often focuses on spatial extents that are much larger than those traditionally studied in ecology, often, the landscape as seen by a human observer" (Turner *et al.* 2001).	• Turner, M. G., R. H. Gardner, and R. V. O' Neill. 2001. *Landscape Ecology in Theory and Practice: Pattern and Process*. New York: Springer-Verlag.

hierarchical and pluralistic cross-disciplinary framework for promoting interactions and synergies between different perspectives and methods. Finally, the relevance of this framework to the admirable but elusive goal of unification will be discussed.

15.2 Two dominant approaches to landscape ecology

15.2.1 The European approach

The term landscape ecology was coined by the German geographer, Carl Troll (1939), who was inspired especially by the spatial patterns of landscapes captured by aerial photographs and the ecosystem concept put forward by Arthur Tansley (1935). This new field of study was proposed to combine the horizontal–geographical–structural approach with the vertical–ecological–functional approach, in order to meet the needs for geography to acquire ecological knowledge of land units and for ecology to expand its analysis from local sites to the region (Troll 1971). For example, information obtained from local sites through ground-based work can be "extended areally by means of knowledge of the distribution of the ecosystems derived from air photograph study" (Troll 1971). From its very beginning, landscape ecology evidently had a close conceptual relationship with ecosystem ecology. In a formal definition, Troll (1968) described landscape ecology as "the study of the main complex causal relationships between the life communities and their environment in a given section of a landscape. These relationships are expressed regionally in a definite distribution pattern (landscape mosaic, landscape pattern) and in a natural regionalization at various orders of magnitude" (Troll 1968, 1971). While the above definition seems semantically indistinguishable from that of ecosystem ecology, Troll's explanation of the "complex causal relationships" points to three important characteristics that distinguish landscape ecology from ecosystem ecology: (1) broad spatial scales, (2) spatial pattern, and (3) multiplicity of scales.

In addition, a landscape as perceived by Troll (1939, 1971) includes humans in addition to its physical and biological components, as does the ecosystem by Tansley (1935). Like other holistic geographers in Europe and Russia of that time, Troll considered a landscape as something of a *Gestalt* (a German word referring to a configuration of elements or an integrated system organized in such a way that the whole cannot be described merely as the sum of its parts). Zonneveld (1972) further emphasized the holistic totality of the landscape while defining landscape ecology as part of the applied science of land evaluation and planning (Table 15.1). Oddly, he claimed unequivocally that landscape ecology was not part of the biological sciences, but a branch of geography. The holistic landscape perspective culminated in Naveh's and Liberman's (1984, 1994) work which described a landscape as a biocybernetic subsystem of the so-called "Total Human Ecosystem" – "the highest level of co-evolutionary complexity in the global ecological hierarchy" (Naveh 2000). Naveh (1991) further stated that "Landscape ecology deals with landscapes as the total spatial and functional entity of natural and cultural living space. This requires the integration of the geosphere with the biosphere and the noospheric human-made artifacts of the technosphere." This is essentially what is called the "holistic landscape ecology," often described as a transdisciplinary environmental science (Naveh 2000).

In general, most landscape ecological studies in Europe since the 1930s have reflected more of the humanistic and holistic perspective, involving landscape mapping, evaluation, conservation, planning, design, and management (Zonneveld 1972, Naveh and Lieberman 1984, Schreiber 1990, Bastian and Steinhardt 2002). However, it should be pointed out that, influenced by geographic and socioeconomic settings as well as academic and cultural traditions, European landscape ecological studies do vary in terms of the research focus and methodology, ranging from tedious technical mapping of heavily populated areas and systematic land evaluation, to philosophical (and sometimes enigmatic) discourses of the wholeness of landscapes. Some of the fine traditions and exciting new developments in European landscape ecology are well reflected in several chapters of this volume (e.g., Antrop, Chapter 10, Voss *et al.*, Chapter 13, Fry *et al.*, Chapter 14).

15.2.2 The North American approach

Landscape ecology was introduced to North America in the early 1980s (Forman 1981, Risser *et al.* 1984, Forman and Godron 1986), more than 40 years after it had been practiced in central Europe, focusing on the human–land systems. In the following decade, landscape ecology quickly flourished in North America with a stream of new perspectives and methods (Forman 1990, Turner 2005; also see Iverson, Chapter 2 of this volume for an interesting and personable account of the early days of North American Landscape Ecology). Consequently, landscape ecology became a well-recognized scientific discipline around the world by the mid-1990s. In their ground-breaking book, Forman and Godron (1986) defined landscape ecology as the study of the structure, function, and change of landscapes of kilometers wide over which local ecosystems repeat themselves (also see Forman 1995). Landscape structure refers to "the spatial relationships among the distinctive ecosystems"; function refers to "the flows of energy, materials, and species among the component ecosystems"; and change refers to "the alteration in the structure and function of the ecological mosaic over time" (Forman and Godron 1986). This definition of landscape ecology is consistent with Troll's original definition in that both aim to integrate the spatial pattern of landscapes with ecological processes within them. However, Forman and Godron (1981, 1986) provided the first systematic conceptual framework for studying landscape pattern and processes, signified by the patch–corridor–matrix model. As a convenient spatial language, this model has played an important role in promoting the development of landscape ecology worldwide since the 1980s.

Several other definitions of landscape ecology have been developed in North America (see Table 15.1). In particular, the report by Risser *et al.* (1984) was an important landmark publication because it reflected the collective view by North American ecologists on what landscape ecology should be and because it has served as a blueprint for the development of landscape ecology in North America in the past decades. The document is a synthesis of a workshop on landscape ecology held in the USA in April 1983, with 25 participants many of whom were leading ecologists and geographers (23 from the USA, 1 from Canada, and 1 from France). Risser *et al.* (1984) defined landscape ecology as the study of the development, management, and ecological consequences of spatial heterogeneity, or "the relationship between spatial pattern and ecological processes [that] is not restricted to a particular scale." They further identified four "representative questions" in landscape ecology: (1) How does landscape heterogeneity interact with fluxes of organisms, material, and energy? (2) What formative processes, both historical and present, are responsible for the existing pattern in a landscape? (3) How does landscape heterogeneity affect the spread of disturbances (e.g., pest outbreaks, diseases, fires)? (4) How can natural resource management be enhanced by a landscape approach? These earlier ideas of landscape ecology in North American were significantly influenced by the theory of island biogeography (MacArthur and Wilson 1967, Wu and Vankat 1995) and patch dynamics (Levin and Paine 1974, Pickett and White 1985, Wu and Loucks 1995).

In line with Risser *et al.* (1984), the different definitions developed in North America all have considered spatial heterogeneity as the cornerstone of landscape ecology. Of course, this does not mean that all North American landscape ecologists hold the same view on landscape ecology. Their major differences seem to hinge on how a landscape is perceived. In the seminal work of Forman and Godron (1981, 1986), a landscape is a kilometers-wide land area with repeated patterns of local ecosystems (also see Forman 1995). But most landscape ecologists consider landscape simply as a spatially heterogeneous area whose spatial extent varies depending on the organisms or processes of interest (Wiens and Milne 1989, Wu and Levin 1994, Pickett and Cadenasso 1995, Turner *et al.* 2001). In this case, landscape is an "ecological criterion" whose essence is not its absolute spatial scale, but rather its heterogeneity relevant to a particular research question (Allen and Hoekstra 1992, Pickett and Cadenasso 1995).

As such, the idea of "landscape" is also applicable to aquatic systems (Steele 1989, Turner *et al.* 2001, Poole 2002, Wiens 2002, Turner 2005). This multiple-scale or hierarchical concept of landscape is more appropriate because it is consistent with the scale multiplicity of patterns and processes occurring in real landscapes, and because it facilitates theoretical and methodological developments by recognizing the importance of micro-, meso-, macro-, and cross-scale approaches. Today, the most widely used definition of landscape ecology in North America, and arguably worldwide, is simply the study of the relationship between spatial pattern and ecological processes over a range of scales (Pickett and Cadenasso 1995, Turner *et al.* 2001, Turner 2005). Reflective of this dominant ecological paradigm in contemporary landscape ecology are several chapters in this volume, addressing a series of key issues focusing on the interrelationship among spatial pattern, ecological processes, and scale (see Chapters 2 to 9, this volume).

15.3 The elusive goal of a unified landscape ecology

It is evident that the European and North American approaches to landscape ecology have differed historically. On the one hand, the European approach is characterized by a holistic and society-centered view of landscapes, the focus on user-inspired and solution-driven research, and the combination of qualitative empirical methods with surveying and mapping techniques. On the other hand, the North American approach is dominated by an analytical and biological ecology-centered view of landscapes, the focus on basic science-oriented and question-driven studies, and the emphasis on the use of quantitative methods (particularly spatial pattern analysis and modeling). This dichotomy, of course, is an oversimplification of the reality because neither of the two approaches is internally homogeneous in perspectives and because both have been changing as an inevitable consequence of increasing communications and collaborations among landscape ecologists worldwide.

Both European and North American approaches can be traced back to the original definition of landscape ecology by Carl Troll (1939, 1968, 1971). The focus of the North American approach on the interrelationship between spatial pattern and ecological processes is not only consistent with Troll's original definition, but also represents a significant advance in implementing Troll's proposal to integrate the geographical and structural approach with the ecological and functional approach. Also, as noted earlier, the emphasis on large geographic areas, spatial patterns, and scale multiplicity that characterizes the

North American approach was evident in Troll's earlier writings. One may argue that Carl Troll was inspired as much by landscape patterns revealed in aerial photos in the 1930s as contemporary landscape ecologists are by those displayed in GIS. Indeed, it was the resurgence of interest in linking ecological processes with spatial pattern in the 1980s that led to a revitalization of the entire field of landscape ecology. Studies of spatial heterogeneity have laid an important foundation for landscape ecology as a scientific enterprise. On the other hand, landscape ecological studies in Europe have epitomized the ideas of landscapes as human-dominated *gestalt* systems, which were also evident in the early works of Troll and other holistic landscape ecologists (Troll 1971, Naveh and Lieberman 1984, Bastian and Steinhardt 2002). They have promoted the development of interdisciplinary and transdisciplinary approaches that transcend natural and social sciences. Undoubtedly, these studies provide valuable methods and exemplary solution strategies for dealing with various complex landscape issues, which must also be considered as an integral part of landscape ecology.

The simplistic dichotomy of landscape ecology approaches also obscures the fact that North American landscape ecology has recognized the important role that humans may have in shaping landscapes from its very beginning. In most cases, humans have been treated as "one of the factors creating and responding to spatial heterogeneity" (Turner *et al.* 2001, Turner and Cardille, Chapter 4, this volume), but perspectives from landscape architecture and planning are quite prominent in other instances (e.g., Nassauer 1997, Ahern 1999, Vos *et al.*, Chapter 13, this volume). In contrast, human society becomes the focus in European landscape ecology as presented by Naveh and Lieberman (1984, 1994). While advocating this holistic landscape ecology perspective, Naveh (1991) claimed that North American landscape ecology was merely "a ramification and spatial expansion of population, community, and ecosystem ecology," and that Risser *et al.*'s (1984) vision of landscape ecology as "the synthetic intersection of many related disciplines which focus on spatial and temporal pattern of the landscape" was inadequate. However, although the North American approach does not always consider landscapes in "their totality as ordered ecological geographical and cultural wholes," even the most ardent holists cannot deny that studies using this approach "are important and of great theoretical and epistemological value to the science of landscape ecology" (Naveh 1991). On the other hand, few would doubt that a holistic landscape ecology approach is essential for resolving problems of biodiversity conservation and ecosystem management.

During the past decade, there have been an increasing number of books and articles attempting to unite the two primary approaches to landscape ecology (Farina 1998, Wiens 1999, Bastian 2001, Wu and Hobbs 2002, Burel and

Baudry 2003). While landscape ecologists converge on the desire for a unified landscape ecology, they differ significantly as to how to achieve the goal. How can different perspectives be unified? There is no simple way to add them up to form a coherent scientific core of landscape ecology even if such a "core" exists. One common approach that many ecologists have adopted is to include humans and their activities as factors influencing and responding to landscape heterogeneity. In this case, landscape ecology is viewed as a branch of ecology, and issues of land use, biodiversity conservation, ecosystem management, and landscape planning and design belong to the domain of practical applications of landscape ecology, or "applied landscape ecology" (Turner *et al.* 2001).

Others do not seem to agree. For example, Naveh (1991) asserted that "landscape ecologists cannot restrict themselves merely to the study of the ecology and/or geography or history of landscapes, projected according to the definition of Forman and Godron (1986)," and that "landscape ecological studies have to be carried out along multidimensional, spatio-temporal, functional, conceptual and perceptional scales by multidisciplinary teams, using innovative interdisciplinary methods and having a common systems approach and transdisciplinary conception of landscape ecology." We agree that interdisciplinarity and transdisciplinarity are critically important to landscape ecology (Wu and Hobbs 2002), and this point has been made clear and loud in most of the chapters of this volume (e.g., Hof and Flather, Chapter 8, Mackey *et al.*, Chapter 11, Bowman, Chapter 12, Fry *et al.*, Chapter 14). However, we do not believe that each and every landscape ecological study has to be done "along multidimensional, spatio-temporal, functional, conceptual and perceptional scales by multidisciplinary teams." Interdisciplinarity and transdisciplinarity are not monolithic, but hierarchical. Thus, we argue that the unification of landscape ecology needs a complementary framework that clearly recognizes and takes advantage of the hierarchical structure in cross-disciplinarity.

15.4 A hierarchical and pluralistic framework for landscape ecology

When a group of leading scientists from around the world was asked about the future of landscape ecology, they unanimously agreed that the field is characterized, most prominently, by its interdisciplinarity or transdisciplinarity (see Wu and Hobbs 2002). It is logical, then, to take this consensus as a point of departure for exploring the possibility of unifying different landscape ecology perspectives. However, we need to understand what landscape ecologists mean by the terms interdisciplinarity and transdisciplinarity because they have been used rather ambiguously in the literature. Particularly, transdisciplinarity sometimes sounds like "a mystic supra-paradigm" that can hardly be understood in practical terms, much less implemented (Tress *et al.* 2005). Thus, we believe that clearly defined terms for cross-disciplinary interactions are a prerequisite for effective discussions on the possible unification of landscape ecology approaches.

Based on an extensive review of the literature, Tress *et al.* (2005) and Fry *et al.* (Chapter 14, this volume) have provided a much needed clarification on four frequently used terms with increasing degrees of cross-disciplinary integrations: disciplinarity, multidisciplinarity, interdisciplinarity, and transdisciplinarity. Disciplinary research operates within the boundary of a single academic discipline with no interactions with other disciplines, thus producing disciplinary knowledge; multidisciplinary research involves two or more disciplines with loose between-disciplinary interactions and a shared goal but parallel disciplinary objectives, thus producing "additive" rather than "integrative" knowledge; interdisciplinary research involves multiple disciplines that have close cross-boundary interactions to achieve a common goal based on a concerted framework, thus producing integrative knowledge that cannot be obtained from disciplinary studies; and transdisciplinary research involves both cross-disciplinary interactions and participation from nonacademic stakeholders or governmental agencies guided by a common goal, thus producing integrative new knowledge and uniting science with society (Tress *et al.* 2005, Fry *et al.*, Chapter 14, this volume). According to these authors, both interdisciplinary and transdisciplinary, but not multidisciplinary, studies are "integrative" research, and transdisciplinarity is essentially interdisciplinarity plus nonacademic involvement. Of course, disciplines or sub-disciplines are relative and dynamic terms that depend necessarily on the classification criteria used. Thus, it is important to recognize that cross-disciplinarity (i.e., multi-, inter-, and transdisciplinarity) may be discussed in different domains, such as within biological sciences, among natural sciences, or across natural and social sciences.

Before we discuss our cross-disciplinary framework for landscape ecology, let's make some general observations of the science of ecology first. Ecology has often been described as an interdisciplinary science because the relationship between organisms and their environment involves a myriad of biological, physiochemical, and geospatial processes. Thus, ecological concepts, theories, and methods come from a number of different disciplines, including botany, zoology, evolutionary biology, genetics, physiology, soil science, physics, chemistry, geography, geology, meteorology, climatology, and remote sensing. Without a common ecological context, some of these disciplines may seem rather unrelated. Various interactions among these disciplines characterize different ecological sub-disciplines (e.g., molecular ecology, chemical ecology, physiological ecology, ecosystem ecology, geographical ecology, etc.). Arguably, the most popular way of classifying ecological sub-disciplines, at least among bio-ecologists, has been based on the hierarchical levels of biological organization from the organism to population, community, ecosystem, landscape, and the biosphere. Although this is not a nested hierarchy (meaning that the levels do not always correspond to spatial and temporal scales in a consistent order), some general patterns of cross-disciplinarity emerge along the hierarchy.

Moving up the hierarchy of biological organization from physiological ecology at the level of individual organisms to global ecology that focuses on the entire Earth system, research questions and methodologies, in general, become increasingly multidisciplinary and interdisciplinary, spatial and temporal scales characterizing each field tend to increase, and mechanistic details of phenomena under study tend to get increasingly coarse-grained. The need and actual frequency of explicitly considering human activities in research also tend to increase. For example, interdisciplinary studies that involve both natural and social sciences are much more frequently encountered in ecological studies at the landscape and global levels than those focusing on individual organisms and local biological communities. As different ecological disciplines provide different perspectives and approaches to the study of nature, they all contribute crucial knowledge to understanding how nature works in the multiscaled and diversely complex world. Generally, studies at lower levels of the ecological hierarchy provide the mechanisms for patterns observed at higher levels, whereas higher-level studies provide the context and significance for lower-level processes. For instance, it is impossible to understand how terrestrial biomes respond to global climate change without invoking the knowledge of plant ecophysiology and ecosystem ecology. On the other hand, global climate change has provided tremendous impetus and new directions for physiological and ecosystem ecology.

The above general patterns suggest that the interdisciplinarity of ecology is quite heterogeneous. We argue that landscape ecology has similar disciplinary characteristics in that landscape ecology involves essentially all the levels of ecological organization and as diverse disciplines as ecology itself. Although the landscape sometimes is considered as a level of ecological organization, it is fundamentally a hierarchical concept that is operational on a wide range of scales in space and time. Different from the traditional ecological disciplines, landscape ecology focuses explicitly on the relationship between spatial pattern and ecological processes on the one hand and nature–society interactions on the other, with the human landscape as arguably the most common scale of research activities.

To promote synergies and unification in the extremely heterogeneous field of landscape ecology, we argue that interdisciplinarity and transdisciplinarity should be interpreted in a hierarchical and pluralistic view (Fig. 15.1).

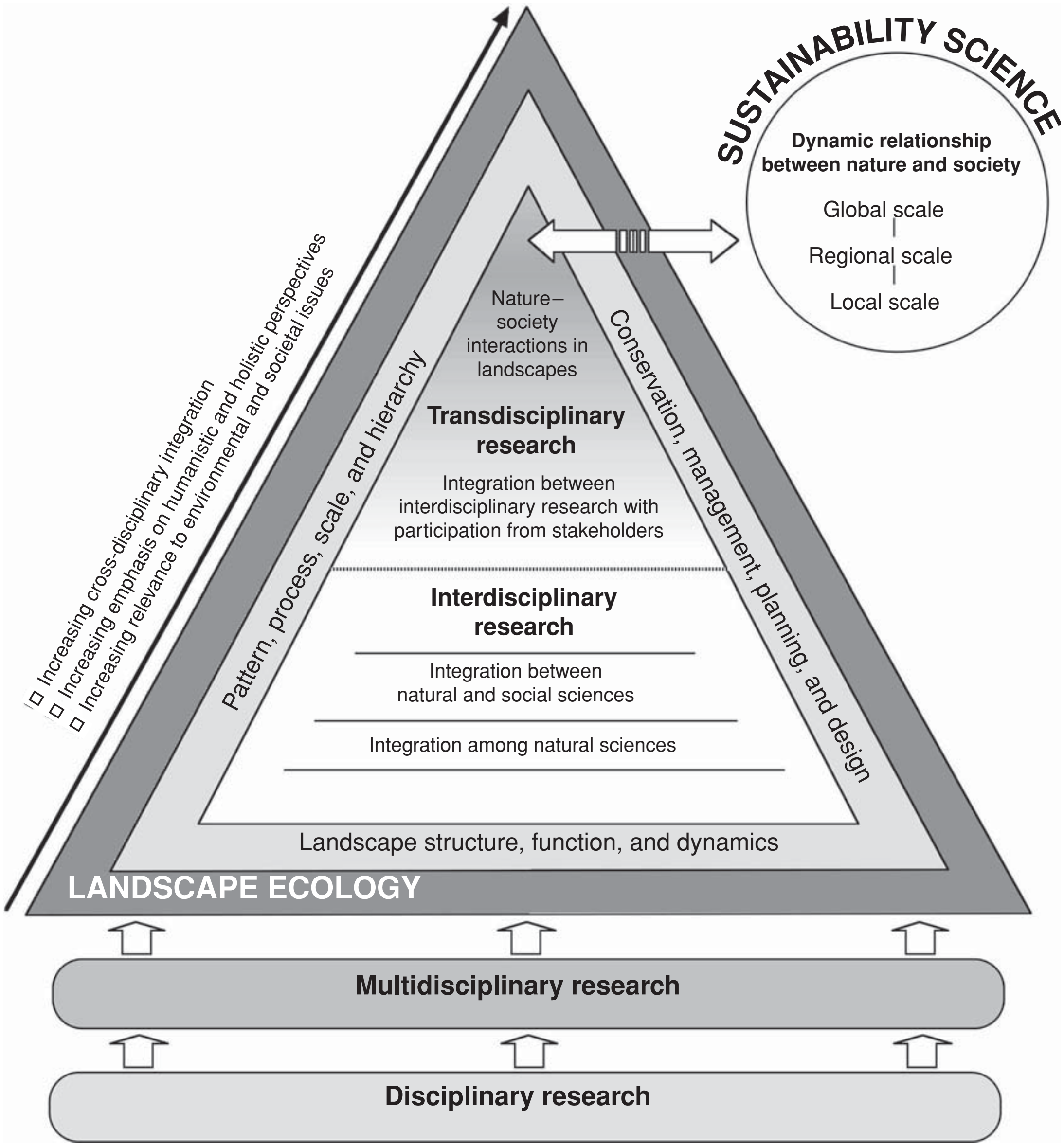


FIGURE 15.1
A hierarchical and pluralistic view of landscape ecology as an interdisciplinary and transdisciplinary science. Landscape ecology is composed of research with various degrees of cross-disciplinary integration from interdisciplinary studies involving multiple natural sciences (e.g., bio-ecology and physical geography) to transdisciplinary studies that include natural and social sciences as well as active participation by stakeholders. Relevant multidisciplinary and disciplinary studies can also provide important contributions to the science of landscape ecology. The definitions of cross-disciplinarities are based on Tress *et al.* (2005)

"Hierarchical" here refers to the multiplicity of organizational levels, spatiotemporal scales, and degrees of cross-disciplinary interactions as well as the relativity of the definition of discipline. As a whole, landscape ecology is an integrative science that consists of studies with different degrees of interdisciplinary and transdisciplinary integration. This basic cross-disciplinary structure is not only reflective of what landscape ecology has been, but also germane to its future development. For example, it seems consistent with the general theme emerging from a list of major research directions and challenges suggested by a group of leading landscape ecologists (Wu and Hobbs 2002), as well as the chapters in this volume. In addition, it is hard to imagine how a credible transdisciplinary science can be developed without resorting to interdisciplinary and multidisciplinary efforts as well as solid disciplinary bases. "Pluralistic" here indicates the necessity to recognize the values of different perspectives and place them in a proper context characterized by a hierarchical cross-disciplinarity. This is indispensable for landscape ecology because of its diverse origins and objectives.

In this hierarchical and pluralistic framework, various approaches and perspectives correspond to different levels in the pyramid of cross-disciplinary integration (Fig. 15.1). In reality, landscape ecological studies usually have varying degrees of cross-disciplinary integration that are determined by specific research goals and questions. Many influential landscape ecological studies have involved different degrees of interdisciplinarity concerning primarily natural sciences, such as biological, ecological, physical, and geographical disciplines. The research topics include the effects of landscape pattern on animal behavior or "behavioral landscape ecology," metapopulation dynamics, spread of disturbance across landscapes, spatial ecosystem processes, patch dynamics, and neutral landscape models (e.g., Turner 1989, Farina 1998, Burel and Baudry 2003, Turner and Cardille, Chapter 4, Fahrig, Chapter 5, Gardner *et al.*, Chapter 6, this volume). In general, moving from the bottom to the top of the cross-disciplinarity pyramid in Fig. 15.1, landscape ecology increases the degree of integration among disciplines, prominence on humanistic and holistic perspectives, and relevance to environmental and societal issues (e.g., Hof and Flather, Chapter 8, Ludwig, Chapter 9, Mackey *et al.*, Chapter 11, Bowman, Chapter 12, Vos *et al.*, Chapter 13, this volume). Correspondingly, human–environment interactions increasingly become the focus of landscape ecology towards the transdisciplinarity end. There are outstanding examples from Europe and elsewhere in which natural and social sciences are successfully integrated with direct involvement of stakeholders, policy-makers, and governmental agencies (see Fry *et al.*, Chapter 14, this volume). Such transdisciplinary research ultimately unites science with society, and is an indispensable part of landscape ecology. In this case, landscape ecology is a critical part of the emerging sustainability science that focuses on the dynamic interactions between nature and society from the local to global scale through place-based and problem-driven projects (Kates *et al*. 2001, Clark and Dickson 2003).

15.5 Discussion and conclusions

Landscape ecology is the science and art of studying and influencing the relationship between spatial pattern and ecological processes across hierarchical levels of biological organization and different scales in space and time. The relationship among pattern, process, and scale is as essential in human-dominated landscapes as in natural landscapes, and is as important in theory as in practice. The "science" of landscape ecology focuses on understanding the dynamics of spatial heterogeneity and the relationship among pattern, process, and scale in natural as well as human-dominated landscapes. The "art" of landscape ecology emphasizes the necessary use of humanistic and holistic perspectives for integrating biophysical with socioeconomic and cultural components in general, and design, planning, and management in particular.

As we discussed earlier, two salient approaches have evolved, both of which can be traced back to the original definition of landscape ecology by Carl Troll (1939, 1968, 1971). The pattern–process–scale perspective that characterizes the North American approach is a continuation and indeed a breakthrough of realizing Troll's aspiration to integrate the geographical (structural) and ecological (functional) approaches. On the other hand, inspired and constrained by the close interactions between land and human society, scientists particularly in European and the Mediterranean countries have transformed the early holistic ideas into a transdisciplinary vision for landscape ecology. Differences in perspectives have apparently caused some landscape ecologists to worry about an identity crisis for landscape ecology (e.g., Moss 1999, Wiens 1999), and others have increasingly called for a unification of different approaches to landscape ecology (Wiens and Moss 1999, Bastian 2001, Wu and Hobbs 2002). Nonetheless, landscape ecology has been maturing as a science in recent years as it has apparently become more quantitative and precise with increasing use of modeling and statistical approaches, more concentration on methodology, and more concerted efforts to bring together different perspectives (Hobbs 1997, Wu and Hobbs 2002).

We believe that the diversity, but not divergence, of perspectives is an essential characteristic and strength of landscape ecology. The hierarchical and pluralistic framework proposed in this chapter help unite the different approaches to landscape ecology and allows for the continuing development of diverse perspectives and approaches. Unification is not to make certain views more prominent by diminishing others, but rather to join different perspectives complementarily in order to produce a whole that is larger than the sum of its parts. This is especially true for broadly interdisciplinary and transdisciplinary sciences such as landscape ecology that cut across natural and social sciences. Landscape ecology may never have a monolithic disciplinary core, and it should not in view of its diverse origins and goals. As a science of spatial heterogeneity, landscape ecology can benefit from its disciplinary heterogeneity. On the one hand, landscape ecology will continue to improve our understanding of the relationship among pattern, process, and scale; and on the other hand, it should play an increasingly important role in sustainability science in years to come.

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References

Ahern, J. 1999. Integration of landscape ecology and landscape design: an evolutionary process. Pages 119–23 in J.A. Wiens and M.R. Moss (eds.). *Issues in Landscape Ecology*. Snowmass Village: International Association for Landscape Ecology.

Allen, T.F.H. and T.W. Hoekstra. 1992. *Toward a Unified Ecology*. New York: Columbia University Press.

Bastian, O. 2001. Landscape ecology: towards a unified discipline? *Landscape Ecology* **16**, 757–66.

Bastian, O. and U. Steinhardt (eds.). 2002. *Development and Perspectives in Landscape Ecology*. Dordrecht: Kluwer.

Burel, F. and J. Baudry. 2003. *Landscape Ecology: Concepts, Methods and Applications*. Enfield, NH: Science Publishers, Inc.

Clark, W.C. and N.M. Dickson. 2003. Sustainability science: the emerging research program. *Proceedings of the National Academy of Sciences (USA)* **100**, 8059–61.

Farina, A. 1998. *Principles and Methods in Landscape Ecology*. London: Chapman & Hall.

Forman, R.T.T. 1981. Interaction among landscape elements: a core of landscape ecology. Pages 35–48 in S.P. Tjallingii and A.A. de Veer (eds.). *Perspectives in Landscape Ecology: Contributions to Research, Planning and Management of Our Environment*. Wageningen: Pudoc.

Forman, R.T.T. 1990. The beginnings of landscape ecology in America. Pages 35–41 in I.S. Zonneveld and R.T.T. Forman (eds.). *Changing Landscapes: An Ecological Perspective*. New York: Springer-Verlag.

Forman, R.T.T. 1995. *Land Mosaics: The Ecology of Landscapes and Regions*. Cambridge: Cambridge University Press.

Forman, R.T.T. and M. Godron. 1981. Patches and structural components for a landscape ecology. *Bioscience* **31**, 733–40.

Forman, R.T.T. and M. Godron. 1986. *Landscape Ecology*. New York: John Wiley & Sons, Inc.

Hobbs, R.J. 1997. Future landscapes and the future of landscape ecology. *Landscape and Urban Planning* **37**, 1–9.

Kates, R.W., W.C. Clark, R. Corell, *et al.* 2001. Sustainability Science. *Science* **292**, 641–2.

Levin, S. A. and R. T. Paine. 1974. Disturbance, patch formation and community structure. *Proceedings of the National Academy of Sciences (USA)* **71**, 2744–7.
MacArthur, R. H. and E. O. Wilson. 1967. *The Theory of Island Biogeography*. Princeton: Princeton University Press.
Moss, M. R. 1999. Fostering academic and institutional activities in landscape ecology. Pages 138–44 in J. A. Wiens and M. R. Moss (eds.). *Issues in Landscape Ecology*. Snowmass Village: International Association for Landscape Ecology.
Nassauer, J. I. 1997. Culture and landscape ecology: insights for action. Pages 1–11 in J. I. Nassauer (ed.). *Placing Nature: Culture and Landscape Ecology*. Washington, DC: Island Press.
Naveh, Z. 1991. Some remarks on recent developments in landscape ecology as a transdisciplinary ecological and geographical science. *Landscape Ecology* **5**, 65–73.
Naveh, Z. 2000. What is holistic landscape ecology? A conceptual introduction. *Landscape and Urban Planning* **50**, 7–26.
Naveh, Z. and A. S. Lieberman. 1984. *Landscape Ecology: Theory and Application*. New York: Springer-Verlag.
Naveh, Z. and A. S. Lieberman. 1994. *Landscape Ecology: Theory and Application*, 2nd edn. New York: Springer-Verlag.
Pickett, S. T. A. and M. L. Cadenasso. 1995. Landscape ecology: spatial heterogeneity in ecological systems. *Science* **269**, 331–4.
Pickett, S. T. A. and P. S. White. 1985. *The Ecology of Natural Disturbance and Patch Dynamics*. Orlando: Academic Press.
Poole, G. C. 2002. Fluvial landscape ecology: addressing uniqueness within the river discontinuum. *Freshwater Biology* **47**, 641–60.
Risser, P. G., J. R. Karr, and R. T. T. Forman. 1984. *Landscape Ecology: Directions and Approaches*. Special Publication 2. Champaign: Illinois Natural History Survey.
Schreiber, K.-F. 1990. The history of landscape ecology in Europe. Pages 21–33 in I. S. Zonneveld and R. T. T. Forman (eds.). *Changing Landscapes: An Ecological Perspective*. New York: Springer-Verlag.
Steele, J. H. 1989. The ocean "landscape". *Landscape Ecology* **3**, 185–92.
Tansley, A. G. 1935. The use and abuse of vegetational concepts and terms. *Ecology* **16**, 284–307.
Tress, G., B. Tress, and G. Fry. 2005. Clarifying integrative research concepts in landscape ecology. *Landscape Ecology* **20**, 479–93.
Troll, C. 1939. Luftbildplan and okologische bodenforschung. *Zeitschraft der Gesellschaft fur Erdkunde Zu Berlin*, 241–98.
Troll, C. 1968. Landschaftsokologie. Pages 1–21 in *Pflanzensoziologie und Landschaftsokologie – Symposium Stolzenau*. Junk: The Hague.
Troll, C. 1971. Landscape ecology (geoecology) and biogeocenology – a terminological study. *Geoforum* **8**, 43–6.
Turner, M. G. 1989. Landscape ecology: the effect of pattern on process. *Annual Review of Ecology and Systematics* **20**, 171–97.
Turner, M. G. 2005. Landscape ecology in North America: past, present, and future. *Ecology* **86**, 1967–74.
Turner, M. G., R. H. Gardner, and R. V. O'Neill. 2001. *Landscape Ecology in Theory and Practice: Pattern and Process*. New York: Springer-Verlag.
Wiens, J. A. 1992. What is landscape ecology, really? *Landscape Ecology* **7**, 149–50.
Wiens, J. A. 1999. Toward a unified landscape ecology. Pages 148–51 in J. A. Wiens and M. R. Moss (eds.). *Issues in Landscape Ecology*. Snowmass Village: International Association for Landscape Ecology.
Wiens, J. A. 2002. Riverine landscapes: taking landscape ecology into the water. *Freshwater Biology* **47**, 501–15.
Wiens, J. A. and B. T. Milne. 1989. Scaling of "landscape" in landscape ecology, or, landscape ecology from a beetle's perspective. *Landscape Ecology* **3**, 87–96.

Wiens, J.A. and M.R. Moss (eds.). 1999. *Issues in Landscape Ecology*. Snowmass Village: International Association for Landscape Ecology.

Wu, J. and R. Hobbs. 2002. Key issues and research priorities in landscape ecology: an idiosyncratic synthesis. *Landscape Ecology* **17**, 355–65.

Wu, J. and S.A. Levin. 1994. A spatial patch dynamic modeling approach to pattern and process in an annual grassland. *Ecological Monographs* **64**(**4**), 447–64.

Wu, J. and O.L. Loucks. 1995. From balance-of-nature to hierarchical patch dynamics: a paradigm shift in ecology. *Quarterly Review of Biology* **70**, 439–66.

Wu, J. and J.L. Vankat. 1995. Island biogeography: theory and applications. Pages 371–9 in W.A. Nierenberg (ed.). *Encyclopedia of Environmental Biology*. San Diego: Academic Press.

Zonneveld, I.S. 1972. *Land Evaluation and Land(scape) Science*. Enschede, The Netherlands: International Institute for Aerial Survey and Earth Sciences.

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