Institutional Dimension of Biodiversity
Conservation

Ilva Sporne
LLM (University of Latvia), MBA (Riga Technical University),
MEnvMngm (Hons) (Griffith University)

Griffith School of Environment
Griffith University

Submitted in fulfilment of the requirements of the degree of Doctor of Philosophy

June 2014
Abstract

This thesis makes a contribution to the growing body of literature examining the institutional dimension of human-environment interactions. It has been guided by an interest in the problem of loss of terrestrial biodiversity in the state of Queensland, Australia and its institutional determinants. The study explored two research questions:

- How to conceptualise and evaluate the effectiveness of institutions contributing to the resolution of environmental problems?
- How effective is the Queensland land use planning and development assessment system in achieving biodiversity protection outcomes?

The first part of the study established a theoretical and analytical foundation for the effectiveness assessment of institutional environmental performance, by examining a wide range of theoretical, conceptual and analytical questions regarding the conceptualisation of institutions, their causal role and evaluation. The study was built on an understanding of institutions as systems of rules that structure social interactions, and it defined institutional ‘performance’ as an institutional influence on, or contribution to, the behavioural response of targeted actors. It argued that institutions play a significant role in social interactions, and are an important explanatory factor for many behavioural phenomena.

Building on the literature review, the study established that biodiversity protection is a highly complex and multi-faceted problem. Institutional designs are required to address a range of problem attributes, such as the existing knowledge base, value and incentive systems, distribution of decision-making authorities, and coordination of interactions among a large number of actors. In this context, the study examined two analytical problems. The first was how to approach a large diversity of problem attributes that may contribute to the resolution or creation of complex environmental problems. The second was how to examine diverse and complex institutional designs.

The major outcome of the first part of the study was a diagnostic framework that links diverse determinants of institutional environmental performance into one coherent structure. It has been designed by combining the institutional diagnostics approach proposed by Young (2002) with the typology of rules of the Institutional Analysis and Development framework proposed by Ostrom and Crawford (2005). The main purpose of the framework is to provide a diagnostic tool, which assists analysts with selecting, mapping and evaluating configurations of rules that can operate as determinants of environmental outcomes in specified settings.

The second part of the study evaluated the potential of the Queensland land use planning and development assessment system (Queensland Planning System) to halt biodiversity decline in the state. It approached the analysis from two perspectives. From the broader perspective, the
Queensland Planning System was viewed as a complex interacting system of rules established at different governance levels, which cumulatively determine the current land use pattern in Queensland. From the narrower perspective, the study examined the Sustainable Planning Act 2009 (Qld) (SPA) as a separate sub-system of rules regulating the particular scope of land use planning and development assessment decisions. Environmental performance of the SPA was linked to the provisions of the Vegetation Management Act 1999 (Qld), which is the major determinant of the extent of native vegetation cover. The evaluation was designed as a desktop study and data has been gathered by means of document analysis. It followed the structure of the diagnostic framework established in the first part of the study.

The study found that the Queensland government has introduced several potentially effective regulatory responses to reduce development pressures on the natural environment. At the same time, significant problems remain if Queensland is going to halt biodiversity decline. Regulatory responses have only reduced vegetation clearing. Land is a valuable resource, which is required to support economic growth, and finding the balance between prosperity and environmental goals has proven to be difficult. The recent shift in political direction in the Queensland government set a new trajectory for regulation by weakening the protective strengths of a range of regulatory instruments in favour of agricultural development and resource extraction. It can be predicted that, at least in the short term, the effectiveness of institutional responses will decrease. An increase in vegetation clearing, and a loss of remnant vegetation in agricultural areas, is likely to continue. The future trajectory will also depend on the fluctuations in the political environment and type of values society assigns to biodiversity.
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<td>BDWG</td>
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<td>OECD</td>
<td>Organization for Economic Co-operation and Development</td>
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<td>QCMD</td>
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<td>WCED</td>
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<td>WRI</td>
<td>World Resource Institute</td>
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Statement of originality

This work has not previously been submitted for a degree or diploma in any university. To the best of my knowledge and belief, the dissertation contains no material previously published or written by another person except where due reference is made in the dissertation itself.

_______________________
Date: 26.06.2014
Acknowledgments

I wish to acknowledge and thank everyone who helped me in the preparation of this thesis. I am deeply thankful to Emeritus Professor Patricia Dale, my principal supervisor, for her ongoing encouragement and support of all the twists and turns in the directions my research took during the years of study, and for her great patience and care. I also want to thank Pat for introducing me to the world of ecologists that I always wanted to belong to. I thank Professor Darryl Low Choy, my associate supervisor, for his critical insights and introduction to the complexities of strategic planning and Australian politics. This greatly assisted me in re-directing this study and approaching some of my initial ideas more critically.

I acknowledge the assistance of the School of Environment and Griffith Postgraduate Award Scholarship and Griffith University International Postgraduate Research Scholarship.

I also want to thank all researchers who invited me to participate in their projects. I thank Dr Wendy Steele for giving me an opportunity to explore cross-boundary governance problems, and Professor Marcus Sheaves and Dr Rodrigo Bustamante for giving me an opportunity to explore climate change adaptation issues in estuarine and coastal fisheries. I thank Dr John Knight and my fellow PhD students, Raymonde, Kat and Alicia, for allowing me to assist in their science projects. My special thanks to Alex and Iris for being my day to day support in the office.

I express my deep gratitude and appreciation to Māra Siksna and Dr Arnis Siksna for sheltering, supporting and taking care of me, and being my Australian family for all these years. I acknowledge Māra for her assistance in editing and commenting on my thesis drafts, and Arnis for his assistance in the final editing of the text. I also thank all my great friends in Latvia and Australia, and the supportive Brisbane Latvian community, who made me feel at home.

Special thanks are dedicated to my mum, dad and my brother in Latvia, for their great support and care, and for coping with me being away for such a long time. Thank you for always being there for me!
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Chapter 1 - Introduction: Institutions and the Environment

1.1 Introduction

This thesis contributes to the growing body of literature devoted to the analysis of the institutional dimension of environmental problems. The initial problem that led to the evolution of this study was the continuing decline of biodiversity in Australia reported in *Australia State of the Environment 2006* (Beeton et al. 2006). *Australia’s Fourth National Report to the United Nations Convention on Biological Diversity* (Australian Government 2009:4) among the seven major current and long term threats to biodiversity identified ‘the loss, fragmentation and degradation of habitat’ and ‘population growth and unsustainable development’. These issues triggered the question regarding the effectiveness of established institutional systems to address development impacts on the natural environment. Of particular interest for this study was the land use planning and development assessment system operating in the state of Queensland.

Institutions shape the patterns of social interactions and are the means through which societies achieve collective goals (Goodin 1998a). Continuous design, adjustment and change of laws, regulations, agreements, and development of new behavioural patterns are processes that form an integral part of any society. The rapid decline of environmental quality worldwide and the loss of environmental resources and biodiversity have highlighted the need for institutions that can effectively structure complex human-environment interactions (Young 2002). However, the questions of what institutions are ‘effective’ and how the effectiveness can be determined are far from answered.

While the study started with a predefined question, gaps identified in the theoretical and analytical literature led to the gradual expansion of the study. The initial attempt to address the question of institutional effectiveness led to a chain of further questions: *What role do institutions play in the achievement of biodiversity protection outcomes? How to conceptualise institutional effectiveness and effectiveness of institutional environmental performance? What are the core determinants of institutional environmental performance? How should they be measured?* The search for the answers to these and related questions significantly expanded the scope of this study. As a result, it evolved to address two overarching questions:

- How to conceptualise and evaluate the effectiveness of institutions contributing to the resolution of environmental problems?
- How effective is the Queensland land use planning and development assessment system in achieving biodiversity protection outcomes?

This chapter introduces the major themes discussed further in this study. It starts with an introduction to the key concepts and major theoretical and analytical problems. Then it briefly describes the conceptualisation and evolution of biodiversity protection as a policy and management problem. Further, the chapter outlines the role of formal or statutory institutions in
environmental governance and explains the rationale for the evaluation of the Queensland land use planning and development assessment system. The chapter concludes with a brief description of the research process and the structure of this study.

1.2 Institutions and institutional effectiveness

The term ‘institution’ appears frequently in academic and political literature. However, its meaning varies. In the conceptualisation of institutions this study builds on the theoretical platform of New Institutionalism which offers a broad umbrella for a range of disciplinary perspectives on institutional determinants of human actions. Therefore, as a starting point, ‘institutions’ are described as ‘systems of rules’ or ‘rules of the game’ that structure human interactions (North 1990, Hodgson 2006). As social constructs institutions are not separate actors, but serve as the means ‘whereby transactions between individuals, groups and states are mediated and made tolerably predictable’ (Dovers & Hezri 2010:212).

Institutions possess a range of characteristics having implications for their evaluation. They differ in complexity ranging from the rules guiding interactions between few individuals up to complex multi-level systems structuring behaviour of a large variety of actors with diverse roles and powers. They vary in the scope of regulated activities, geographical scale, degree of formalisation, stage of development, as well as in their interactions with other institutions (Young 2002). They can be formally designed ‘products’ or ‘outputs’ of policy-making processes or simply evolve over time as a result of social interactions (North 1990). As complex systems, institutions can produce both intended change in human behaviour, as well as unpredicted effects or consequences (Underdal & Young 2004). Furthermore, they are not designed in isolation but are ‘fitted into’ particular institutional, economic, sociocultural and biophysical contexts (Ostrom 2005).

That the ‘rules of the game’ are important elements of social systems can be hardly disputed, yet an understanding of this complex phenomenon is still evolving (Ostrom 2005). At the level of theory, New Institutionalism has produced a wide range of theoretical insights into institutional determinants of human choice. The studies exploring or theorising on the significance, functions and operation of institutions span a wide range of disciplines addressing both micro- and macro-levels of social order (Goodin 1998b, Scott 2001). At the same time, while several influential reviews have been provided to advance an understanding of institutions (e.g. Hall & Taylor 1996, Goodin 1998b, Peters 1998, Scott 2001), a common theoretical platform explaining institutional roles is still lacking.

Inquiry into whether a particular institution can be regarded as ‘effective’ raises questions of causality, performance and measurement. Therefore, chapter 2 covers several topics related to institutional theory and analysis, including how to:

- reconcile different theoretical perspectives on institutional functions,
- conceptualise institutional effectiveness and effectiveness of institutional environmental performance,
- distinguish institutional effects from the other drivers of behavioural and environmental change,
- select the elements of assessment design.

### 1.3 Biodiversity protection and institutions

The protection of the natural environment outside specifically designated areas has a relatively short institutional history. The crucial turning point, leading to the incorporation of environmental concerns in regulation of development activities, occurred with the *Report of the World Commission on Environment and Development* 1987 (WCED) also called the Brundtland Report. It introduced the concept of ‘sustainable development’, defining it as ‘development that meets the needs of the present without compromising the ability of future generations to meet their own needs’ (WCED 1987:8). Subsequently, the concept has been widely presented as consisting of three interdependent pillars or the so-called ‘triple bottom line’: economic development, social development and environmental protection.


The concept of ‘biodiversity’ was introduced by the scientific community in the 1980’s (Jeffries 2006). However, the Biodiversity Convention and supporting agreements can be regarded as the main drivers of a shared understanding of the concept and its framing as a policy problem. Article 2 of the Convention defines ‘biological diversity’ as:

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

The Biodiversity Convention recognises intrinsic and use values of biological diversity and its importance ‘for evolution and for maintaining life support systems of the biosphere’ (Preamble). It pursues three core objectives: conservation, sustainable use and equitable sharing of benefits arising out of the use of genetic resources (Article 1). Among other obligations, the contracting parties are required to make necessary changes in existing institutional systems, develop new institutional mechanisms and undertake actions to support the achievement of established goals within their jurisdictions. Australia ratified the Biodiversity Convention in 1993.
Biodiversity protection, involving both conservation and sustainable use objectives, is a complex policy problem requiring a wide spectrum of institutional responses. They range from the protection of rare and endangered species up to the inclusion of biodiversity as one of the factors that needs to be ‘balanced’ or ‘traded-off’ against development goals. With the publication of the *Millennium Ecosystem Assessment* (MEA) (MEA 2005a), particular emphasis has been placed on different use values of biodiversity. As a property of ecological systems, biodiversity is recognised as a crucial determinant of systems’ resilience to disturbances. Thus, biodiversity enables sustained provision of a range of ecosystem services important for human well-being (MEA 2005a).

The biodiversity protection agenda has evolved together with major shifts in understanding of the functioning of ecological systems, as well as with changes in environmental governance structures and strategies. The new management paradigm, described as ‘ecosystem’, ‘environmental’ or ‘integrated resource’ management, has produced shifts towards more holistic, integrated and systems approaches to the management of natural resources (Cortner & Moote 1998). Adaptive management, which supports learning oriented experimentation (Holling 1978, Walters 1986) has gained widespread political acceptance as an approach to address the uncertainty inherent in the management of dynamic ecological systems. In the governance systems shifts have been made towards participatory, multi-level and scale-adapted organisational structures (Folke et al. 2005), as well as resource markets (Dovers 2001a).

The introduction of the biodiversity protection problem in the policy-making arena has raised the question of what implications it has for the (re)design and operation of established institutional systems. In the literature, the problem has been approached from different perspectives, ranging from the development of new theories, concepts and models up to proposals on how to restructure complex governance systems. While the need for institutional change is the common theme, systematic examination of the institutional dimension of this problem has not been conducted.

Drawing on the review of the environmental science, management and governance literature chapter 3 seeks to clarify what role institutions play in addressing the biodiversity protection problem and what criteria characterise an ‘effective’ institutional design.

1.4 Statutory systems and their environmental performance

Interest in the institutional dimension of environmental change has evolved in several areas of environmental governance studies. Since the publication of Hardin’s famous essay *The Tragedy of the Commons* in 1968 (Hardin 1968), a large body of research has been conducted worldwide on informal institutions organising small scale, common pool resource (CPR) management systems such as fisheries, forestry, farming and irrigation (Ostrom 1990, Ostrom et al. 1994, Dietz et al. 2003). Significant advances in understanding the performance of international environmental regimes have been made by the group of scholars involved in the Institutional

The recent studies have made a significant contribution to the understanding of institutional roles in structuring human-environment interactions. However, the literature contains two major gaps. Currently the analysis is predominately concerned with so called ‘environmental’ institutions designed to manage particular types of interactions such as pollution, resource extraction, resource or ecosystem conservation or preservation. At the same time, as the analysts acknowledge (Mitchell 2008, Underdal 2008), significant gaps still remain in the evaluation of environmental effects of ‘non-environmental’ institutions designed to regulate diverse development activities.

Another gap remains in the evaluation of formal statutory systems. With increased interest in new community-based resource governance strategies, attention has been paid to the operation of informal institutional mechanisms. The studies have examined community-based knowledge systems, self-organisation and informal networks developed to govern complex and dynamic ecological systems (Folke et al. 2002, Olsson et al. 2004, Lebel et al. 2006, Gunderson & Light 2006). Similarly, in the analysis of common-pool resource management arrangements in developing countries formal rules have been largely disregarded as causal factors of behavioural change and often considered as ‘rules on paper’ (Ostrom 2005). As a result, in the academic literature evaluations of existing statutory systems operating at the state scale are relatively scarce and are primarily conducted by the legal discipline.

To this end, this study builds on the argument that formal institutional systems such as statutes form an important part of environmental governance. Therefore, understanding how and to what extent current democratic societies address complex environmental problems such as biodiversity loss requires ‘bringing statutes back in’ the environmental governance debate. The terms ‘formal institutions’ or ‘statutory systems’ as used in this study, however, do not imply a narrow set of enforceable rules issued to control or prescribe certain behaviour. In practice, such policy instruments as laws and regulations often form an overarching framework which triggers the evolution of a large sub-system of rules providing for more specific regulation. In other words, statutory systems can incorporate a large variety of rules ranging from ‘how to’ instructions, giving rise to particular management practices, to overarching normative principles guiding the interpretation of rules.

Formal institutional systems are complex and diverse. Current methodological approaches, theories and frameworks offered by the legal discipline are insufficient for examining their
environmental performance. Therefore the core question addressed in chapter 4 is how to link the different determinants of institutional environmental performance into some coherent structure and to conduct a systematic analysis of diverse institutional designs.

1.5 Land use planning and biodiversity

Land use planning can be described as a process that ‘involves the systematic assessment of the potential of available environmental resources, economic and social conditions to select and adopt particular land use options’ (FAO 1996, n.pag). From an institutional perspective, a land use planning system is a policy instrument applied to regulate the development and use of the land within particular jurisdiction. It comprises a complex regulatory framework covering different aspects of the decision-making process, ranging from establishing land use rights and values up to setting strategic policy directions for the regional and state development.

Regulation of land use planning is not new. For several centuries planning rules have formed an integral part of many governance systems worldwide (Home 1997). However, environmental planning has a relatively short history. The need for broader regulation of development effects on the environment appeared on the international arena in the 1970s. In 1972 the United Nation’s Conference on Human Environment (Stockholm Conference) adopted a Declaration outlining 26 guiding principles for the preservation and enhancement of the human environment. In particular, the Stockholm Declaration proclaimed human responsibility ‘to safeguard and wisely manage the heritage of wildlife and its habitat’ (Principle 4) and placed emphasis on the role of planning which ‘must be applied to human settlements and urbanization with a view to avoiding adverse effects on the environment’ (Principle 15).

Since the 1970s, environmental issues have become an integral part of land use planning and development assessment decisions in Australia (Conaher & Conaher 2000). In the 1990s along with the ‘ecologically sustainable development’ agenda such management outcomes as protection of ecosystems, ecological processes and biodiversity conservation have been incorporated in planning legislation (Stein 2008). The expanding role of planning institutions in environmental governance has raised questions regarding various aspects of their environmental performance. Their regulatory scope and capacity has been discussed in different contexts such as coastal management (Thom 2004), urban biodiversity (Hodgman 2004, Moroney & Jones 2006, Byrne et al. 2010), local government capacity and motivation (Mamouney 2000, Wild River 2003), climate change adaptation (Boer 2010, Abel et al. 2011) and linkages with other institutional arrangements (Gurran 2005, Lowe et al. 2006, Lockwood et al. 2008). Furthermore, several practical guidelines have been produced on biodiversity planning (Fallding et al. 2001, Fallding 2004, Rhodes et al. 2008). Although the effectiveness of these systems in achieving biodiversity protection outcomes has been questioned (Powers 2000), a systematic and comprehensive evaluation is lacking.
This study focuses on the analysis of the land use and development assessment system operating in the state of Queensland, Australia (subsequently called the Queensland Planning System), particularly its effectiveness in dealing with biodiversity. Australia has a federal system of government. According to the separation of powers between the Commonwealth and the state and territory governments under the Commonwealth of Australia Constitution Act 1900 the management of land use planning activities falls under the jurisdiction of the states and territories. As a result, there are eight separate and differently ‘crafted’ systems governing land use planning and development assessment activities. Each system is complex and consists of different configurations of regulatory instruments and therefore requires separate evaluation.

The Queensland Planning System has been of particular interest for this study for several reasons. First, the state of Queensland has jurisdiction over development activities which may impact upon highly important biodiversity values, including those acknowledged as World Heritage areas. Second, the coastline of the state is affected by relatively high development pressures, in particular in the southeast corner. This makes planning regulation an important determinant of biodiversity protection outcomes. Finally, the design of the Queensland Planning System is based on a performance-based approach which differs from the other systems in Australia (Wypych et al. 2005, Baker et al. 2006). While performance-based planning was advocated to improve environmental outcomes (Yearbury 1998), only few studies have analysed environmental performance aspects of this system (Brody 2003, Brody & Highfield 2005).

Building on established theoretical and analytical frameworks, chapters 5 and 6 seek to evaluate the effectiveness of the Queensland Planning System in achieving biodiversity protection outcomes. The analysis is approached from two perspectives. From the broader perspective in chapter 5, it is viewed as a complex interacting system of rules established at different governance levels which cumulatively determine the current land use pattern in Queensland. From the narrower perspective chapter 6 examines the regulatory framework established under the Sustainable Planning Act 2009 (Qld) (SPA) as a separate sub-system of rules.

1.6 Introduction to the research process

Institutional effectiveness analysis is related to public policy evaluation (Rossi et al. 2004, Weimer & Vining 2005). In a broader sense, however, this study can be labelled as ‘design research’. While this type of research is usually associated with so called ‘design sciences’ such as architecture, computer science and engineering (March & Smith 1995, Cross 1999), it is gaining increasing recognition in different disciplines of the social sciences in particular in business (Buchanan 1999) and education (Edelson 2002, Collins et al. 2004). According to Buchanan (1999:9) ‘design’ can be defined as ‘the human power of conceiving, planning, and making products that serve human beings in the accomplishment of their individual and common purposes.’ Institutions, that have been created to achieve greater predictability in human interactions, can be approached as one of the ‘products’ of the design process. In practice, design

To better position this study it is useful to distinguish between three perspectives on the research of institutional design. Drawing on Buchanan (1999) they can be labelled as ‘clinical’, ‘applied’ and ‘basic’ (or ‘theoretical’). So called ‘clinical’ studies focus on the investigation of institutional factors operating as determinants of a particular behavioural pattern (e.g. resource consumption) or modes of social interactions (e.g. collaboration, strategic planning process). ‘Applied’ studies are directed towards more general insights into the institutional dimension of different classes of problems (e.g. common pool resource management). In the environmental context, this stream of research is directed towards discovering more general principles of institutional design (Ostrom 1990, Ostrom et al. 1994, Breitmeier et al. 2011). Finally, the ‘theoretical’ perspective is more fundamental. It involves the generation of institutional theory or theory of institutional design, which, in turn, provides an overall foundation for institutional studies (Goodin 1998b).

To this end, this study consists of two major parts. The first part (chapters 2, 3 and 4) aims to provide the theoretical and analytical foundation for analysing the effectiveness of institutional environmental performance. It belongs to the domain of ‘applied’ design research. While chapter 2 aims to reconcile different theoretical perspectives on institutional role, it does not intend to develop institutional theories. The second part ( chapters 5 and 6) analyses the design of particular institutional system and, thus, takes a more detailed (‘clinical’ approach.

The nature of the study is multidisciplinary. To various levels of detail it brings together theoretical platforms, concepts and empirical findings produced by different disciplines and sub-disciplines of both social and environmental sciences. These include policy science, sociology, economics, law, land use planning, ecology and landscape ecology.

The first part of the study builds on the review of several clusters of academic literature. The theoretical foundation is derived from the work of scholars operating under the umbrella of New Institutionalism, in particular its Rational Choice and Sociological directions. In the development of a conceptual and analytical platform for the analysis it predominately draws on the IDGEC research project literature (see section 1.4), which provides detailed insights into diverse theoretical, conceptual and methodological challenges encountered in the assessment of international environmental regimes. Finally, in examining attributes of ecological systems and biodiversity as science, management and governance problems it reviews a broad spectrum of theoretical and empirical studies produced by both environmental and social sciences.

The major output of the first part of the study is a diagnostic framework. It builds on the classification of rules of the Institutional Analysis and Development (IAD) framework
developed by Ostrom and Crawford (2005). It covers configurations of rules operating at the two hierarchical levels most directly affecting ‘on ground’ environmental outcomes and impacts. It is designed to identify critical dimensions of institutional performance and diagnose their potential to address complex environmental problems such as biodiversity loss. The framework development process, its benefits, limitations and application potential are discussed in detail in chapter 4.

Building on this foundation, the second part of the study (chapters 5 and 6) undertakes an analysis of the Queensland Planning System. Following the structure of the diagnostic framework it unpacks the system into several sub-sets of rules operating at two levels of the governance hierarchy. The assessment is based on a desktop study using data derived from primary and secondary documentary sources. It identifies major problem areas of the Queensland Planning System, which currently, as well as under predicted conditions of increasing development pressures, could contribute to adverse environmental effects, in particular the loss of biodiversity. Detailed discussion and description of employed data sources and methodological approach, as well as related limitations are provided in chapters 5 and 6.

It should be noted that for the sake of clarity this chapter has presented the research as a linear process gradually evolving from the identified research problem to the assessment design and evaluation. In practice, it evolved following a so called ‘interactive model’ (Maxwell 2005). Numerous theoretical, conceptual and analytical problems, as well as gaps in the theoretical and empirical literature, led to ongoing reiteration between all parts of this study. This process has led to several changes in the scope of questions and resulted in a significant shift in research foci. In other words, as described by Maxwell (2005:3), this study ‘involved “tacking” back and forth between the different components of the design, assessing the implications of goals, theories, research questions, methods, and validity threats for one another.’ More detailed reflection on this process is provided in chapter 7, which summarises the major findings and contributions made by this study.

1.7 Thesis structure

This chapter has already introduced the core themes covered by different parts of the study. This concluding section provides a more detailed insight into the structure and the content of each chapter.

Chapter 2 establishes the conceptual and analytical platform of this study. Based on concepts and theories developed under the umbrella of New Institutionalism the first part of the chapter discusses the conceptualisation of institutions and organisations, and identifies the range of functions institutions perform in structuring human interactions. The second part clarifies what is understood by the effectiveness of institutional performance and institutional environmental performance, and describes the core elements of the assessment design. The chapter also
discusses major challenges facing the assessment of institutional environmental performance and resulting implications for assessment design.

Chapter 3 explores the scope and attributes of the biodiversity protection problem, and identifies what implications they may have for institutional design and performance assessment. Addressing the questions - what, where, why, how, and by whom - the chapter places the biodiversity protection problem in science, management and governance contexts. It examines a range of propositions made in the literature regarding how particular attributes of the problem should be addressed. Particular attention is devoted to the concept of adaptive management, which has been widely promoted as the major approach for dealing with complex and unpredictable environmental problems. This chapter forms the basis for the design of the diagnostic framework and subsequent analysis of the Queensland Planning System.

Chapter 4 addresses the question of how to systematically analyse the environmental performance of diverse institutional systems. It discusses the application of institutional diagnostics and the IAD rules framework and describes three major steps in the design of the diagnostic framework. The chapter presents the diagnostic framework for the evaluation of institutional environmental performance and discusses its benefits, limitations and application potential.

Chapter 5 contains the first part of the analysis of the Queensland Planning System. It examines a broad range of institutional factors operating as determinants of land use outcomes in Queensland. It consists of four major parts. The first describes the methodological approach which applies to both parts of the analysis (including chapter 6). The second describes the major drivers of the use of land resources and resulting pressures on the natural environment. The third examines the distribution of regulatory roles and responsibilities between the three levels of government and describes the regulatory system determining environmental outcomes in the state of Queensland. The chapter concludes with a discussion of major strength and weaknesses of the overall regulatory framework in achieving biodiversity protection outcomes. The discussion follows the structure of the diagnostic framework introduced in chapter 4.

Chapter 6 contains the second part of the analysis. It examines the environmental performance of the regulatory framework established under the Sustainable Planning Act 2009 (Qld) (SPA) and consists of four parts. It starts with a brief description of the methodology used for the analysis of the SPA. The second part outlines the evolution of the SPA. The third part describes the structure and instruments of the system with particular focus on provisions of the Vegetation Management Act 1999 (Qld) (VMA) and recent reforms. The chapter concludes with a discussion of the strength and weaknesses of the SPA-VMA framework in achieving biodiversity protection outcomes.

Chapter 7 reflects on the research process and major findings, and outlines the areas for further research. Following the structure of the study the first part reflects on different aspects related to
the development of the theoretical and analytical platforms for the evaluation of institutional environmental performance. The second part discusses the application of the diagnostic approach and synthesises the findings from the evaluation of the Queensland Planning System.

This study employs various concepts and terms requiring definition. While the core concepts have been introduced in this chapter, other definitions used in the study are included in Appendix 1.
Chapter 2 - Institutions, Institutional Effectiveness and Environmental Performance

2.1 Introduction

This chapter introduces the first part of the study guided by the overarching question: how to conceptualise and evaluate the effectiveness of institutions contributing to the resolution of environmental problems.

In both the academic and political literature institutions are increasingly identified as one of the causal factors of environmental and other problems experienced in society. At the same time, the studies exploring institutional effects encounter a number of conceptual and analytical challenges. These arise from the lack of a shared understanding of three major concepts, namely ‘institutions’, ‘institutional performance’ and ‘institutional effectiveness’.

The chapter has two major objectives. First, based on concepts and theories developed under the umbrella of Institutional Theory, particularly New Institutionalism, it seeks to clarify what is understood by ‘institutions’ and what role they play in structuring social interactions (section 2.2). Second, drawing on studies exploring design and performance of environmental institutions, it aims to define what is understood by the effectiveness of institutional performance and institutional environmental performance, and to clarify core elements of assessment design (section 2.3).

2.2 Institutions: what are they and how do they work

Conceptualisation of institutions and their role in shaping human behaviour forms part of an ongoing academic debate spanning different disciplines such as philosophy, sociology, cognitive psychology, economics and policy science. A detailed insight into this debate is beyond the scope of this study (see reviews by Peters 1998, Scott 2001). However, clarification of the core concepts, and an insight into the different theoretical perspectives on institution-actor relations, are essential for establishing the platform for analysis. Drawing on a broad spectrum of the theoretical literature this section addresses two major questions: what are institutions, and how they affect (or are expected to affect) human behaviour?

2.2.1 Institutions, rules and organisations

Differences in conceptualisation of institutions have been identified by many analysts as some of the major challenges for institutional studies (Scharpf 1997, Peters 2000, Ostrom 2005, Hodgson 2006). In both academic and political literature the term ‘institution’ has been used to describe a wide range of phenomena. As introduced in chapter 1, the starting point for this study is the understanding of ‘institution’ as a set of rules that structures human interactions. However, even this definition does not completely resolve differences in the understanding of the concept.
Institutions and rules

This study builds on the platform of Institutional Theory which provides a broad theoretical umbrella for a variety of concepts and theories guiding institutional analysis. According to Scott (2005:461) Institutional Theory ‘considers the processes by which structures, including schemas, rules, norms, and routines, become established as authoritative guidelines for social behaviour’ and ‘inquires into how these elements are created, diffused, adopted and adapted over space and time; and how they fall into decline and disuse’. New Institutionalism or Neo Institutionalism is one of the major directions of Institutional Theory, which currently forms the main theoretical platform for a large number of studies undertaken in various disciplines such as economics, sociology and political science (Scott 2001).

Several schools of thought have been distinguished under New Institutionalism. While individual classifications vary (Hall & Taylor 1996, Peters 1998), they tend to be grouped into two major clusters described as Rational Choice Institutionalism and Sociological Institutionalism (Scott 2001, Young 2002). Both focus on the role of institutions in determining social and political outcomes. At the same time, they support different analytical foci and as Hall and Taylor (1996:936) note, ‘paint quite different pictures of the political world’. According to the theorists significant differences exist in the definition of ‘institution’, as well as in the conceptualisation of institution – actor interactions (Hall & Taylor 1996, Peters 2000, Scott 2001). Moreover, as several scholars (Scharpf 1997, Peters 1998, Scott 2001) argue, a unification of Institutional Theory cannot be achieved and different theoretical directions should be viewed as complementary.

In general, institutional theorists and analysts share a common understanding that the core purpose of institutions is to create greater regularity and predictability of human actions (Peters 2000). For example, Ostrom (2005:3) defines ‘institutions’ as ‘the prescriptions that humans use to organise all forms of repetitive and structured interactions’. Hodgson (2006:2) defines institutions broadly as ‘systems of established and prevalent social rules that structure social interactions’. North (1990:3) defines them as ‘the rules of the game in a society or, more formally [...] the humanly devised constraints that shape human interactions’. Scott (2001:48) describes ‘institutions’ as based on ‘regulative, normative and cultural-cognitive’ elements that ‘provide stability and meaning to social life’.

One of the core differences, however, lies in the understanding of the term ‘rules’ or, as Hodgson (2006:4) notes, ‘as to how far we can stretch the meaning of the term rule in the definition of an institution’. Scott (2001) in his review of various theoretical platforms of New Institutionalism distinguishes between three ‘pillars’ of institutions: regulative, normative and cultural-cognitive. The ‘regulative pillar’ dominates the Rational Choice Institutionalism. Under this direction institutions are explored as regulative systems guiding ‘rule-setting, monitoring, and sanctioning activities’ (Scott 2001:52). The ‘normative’ pillar underpins Sociological Institutionalism which
views institutions more broadly and incorporates values and norms as a source of behavioural change. In this context ‘values’ are understood as conceptions about preferred or desirable outcomes, whereas ‘norms’ specify the appropriate ways of achieving these outcomes. Finally, the Cultural – Cognitive direction\(^1\) describes institutions broadly as ‘shared conceptions that constitute the nature of social reality and the frames through which meaning is made’ (Scott 2008:428). As Scott (2008) points out each ‘pillar’ supports different bases of social order, motives for compliance, logics of actions and institutional mechanisms.

To understand the concept of ‘institution’, and clarify linkages between different sources of behavioural change, this study follows the approach employed by Young (2002). He distinguishes between two broad groups of definitions, described as institutions ‘in the thin sense’ and ‘in the thick sense’. Institutions ‘in the thin sense’ are understood as ‘systems of rules, decision-making procedures, and programs as articulated in constitutive documents’ (Young 2002:5-6). Institutions ‘in the thick sense’ incorporate a broader range of sources including common discourses, informal understandings regarding appropriate behaviour, and routine activities (Young 2002:6).

Following the example, this study distinguishes between the two types of rules. It will use the term ‘formal institutions’ or ‘institutions’ to describe ‘written rules’ or rules that are articulated in constitutive documents such as laws, regulations, agreements, policies, programs and other regulatory instruments. ‘Informal institutions’ or ‘informal rules’ will be used to describe the rules that are embedded in established practices, shared discourses, informal understandings, roles and routines.

Distinguishing between ‘formal’ and ‘informal’ institutions is important for at least two other reasons. First, it implies that the behaviour of institutional actors in any setting can be shaped by two interacting sets of rules. As Ostrom (2005) notes, informal institutions may operate in parallel and either support or create barriers for the implementation of formal rules. Therefore, informal rules may become important explanatory variables for the differences in produced behavioural outcomes in similar settings. Second, on the more fundamental level informal institutions, in the form of embedded norms, shared cognitive concepts and principles, define the platform for the design and implementation of formal rules (Scott 2008). As a result, while formal institutions can be explored as separate systems, the ‘informal’ context also needs to be considered. In other words, the distinction between ‘written’ and ‘unwritten’ rules as two interacting parts of institutional systems can provide a better understanding of institutional determinants of behavioural outcomes.

\(^1\) Note: Cultural-cognitive direction tends to be viewed as part of Sociological Institutionalism (Young 2002).
Another conceptual problem in the theoretical and analytical literature is the distinction between ‘organisations’ and ‘institutions’ (Hodgson 2006). While individuals are considered as the smallest ‘acting unit’, the achievement of institutional outcomes, particularly at higher governance levels, depends on the behaviour of organised groups of individuals or so called ‘collective actors’ (Ostrom 2005). In the literature the term ‘institution’ often has been employed to describe both organisations such as governmental agencies, local governments and corporations, as well as the rules establishing the framework for their operation (Ostrom 2005). Such an application of the term can be partially explained by the nature of organisations which structure interactions of individuals to achieve collective goals. From this perspective, organisations can be described as ‘systems of rules’ or as institutions themselves (Scott 2001, Hodgson 2006). However, as the institutional analysis may require treating organisation as a separate institutional actor (Ostrom 2005), it is important to make the distinction between these two uses of the terms.

Different definitions of ‘organisations’ can be found in the literature. For example, North (1990:5) defines them broadly as ‘groups of individuals bound by some common purpose to achieve objectives.’ Hodgson (2007:96) describes organisations based on three major characteristics: ‘(a) criteria to establish their boundaries and to distinguish their members from non-members, (b) principles of sovereignty concerning who is in charge, and (c) chains of command delineating responsibilities within organisation.’ Furthermore, to distinguish between organisations, or so called ‘corporate actors’ and other collective actors, Scharpf (1997:53) employs a criterion such as the ‘degree of the integration’ of individuals operating within these structures. According to him collective actors are ‘dependent on and guided by the preferences of their members’, whereas composite actors (i.e. organisations) are established to pursue their own goals and are characterised by ‘a high degree of autonomy from the ultimate beneficiaries of their action’ (Scharpf 1997:53).

As an institutional actor an organisation evolves in response to an opportunity provided by institutions and therefore can be seen as institutionally constructed (North 1990, Scott 2005). According to Scharpf (1997:39) institutions define ‘not only the membership of composite actors and the material and legal action resources they can draw upon […] but also the purposes that they are to serve or the values that they are to consider in arriving at their choices.’ The distinction between ‘institutions’ as systems of rules and ‘organisations’ as institutional actors probably has been best described by North (1990:4-5):

The purpose of the rules is to define the way the game is played. But the objective of the team within that set of rules is to win the game – by a combination of skills, strategy, and coordination […]. Modelling the strategies and the skills of the team as it develops is a separate process from modelling the creation, evolution, and consequences of the rules.
Drawing on North (1990) and Scharpf (1997), the study makes a clear distinction between ‘institutions’ as systems of rules and ‘organisations’ as institutionally constructed actors. Therefore, the term ‘institution’ will be employed to describe systems of rules, both formal and informal, that guide interactions of individual and collective actors, including organisations. Systems of rules established within an organisation to structure interactions of its members will be referred to as ‘organisational rules’.

2.2.2 Institutions and institutional actors

The analysis of institutional effectiveness is inevitably linked to the questions regarding the causal significance of institutions (Underdal 2008). New Institutionalism comprises a large range of perspectives on institution-actor relations. Building on different disciplinary platforms, the theorists offer divergent views on the main drivers of human choice. Therefore, it is important to outline different approaches to the conceptualisation of human behaviour employed as foundations of institutional design and analysis. Understanding these differences is also fundamental for the analysis of different approaches taken to the design of environmental policies such as market mechanisms, public involvement or collective decision-making approaches.

All institutional theories are based on implicit or explicit assumptions regarding the behaviour of human actors (North 1990). Most institutional theorists and analysts exploring the design and operation of formal institutions draw on the common assumption of ‘rational’ human behaviour. Rationality implies that the decisions about the preferred course of action are based on reasonable or purposeful evaluation (Kato 1995) or ‘intelligent’ choice (March 1978). At the same time, rationality is not a uniformly defined concept (Scott 2001). The analysts from Rational Choice and Sociological directions of New Institutionalism (see section 2.2.1) build their work on different assumptions regarding the institutional determinants of the two components of human behaviour: perceptions and preferences.

Perception - or ‘deciphering of environment’ (North 1990) or ‘cognitive orientations’ (Scharpf 1997) - broadly describes the cognitive or information processing component of human behaviour. Individuals differ in the ways they perceive and process information. As Scharpf (1997:19) notes, humans act ‘not on the basis of objective reality but on the basis of perceived reality and of assumed cause-and-effect relationships operating in the world they perceive’. Consequently, actions depend on how individual actors perceive the problem, its causes, desirable outcomes, available choices and causal relations between the selected course of actions and outcomes (Scharpf 1997). As North (1990) points out, perception is affected by two major factors: first, the ability of human actors to understand the environment and second, the complexity of problems they have to resolve. In the case of individuals perceptual capacity is determined by cognitive abilities, while organisations ‘depend on interpersonal information processing and communication’ (Scharpf 1997:58).
Behavioural models employed in institutional design and analysis have been built on different assumptions regarding information processing (‘deciphering’) abilities. At one end of the spectrum neoclassical economists have built their market models on the assumption of fully rational, purposeful individuals with stable preferences and abilities to obtain ‘true’ models of the world, or receive information that leads to such models (North 1990, Vatn 2005). They assume that individuals possess ‘a predefined ability to understand both own needs, other’s performance and the working of the natural world’ (Vatn 2005:206). At the other end, cognitive psychologists argue that in real life settings preferences are unstable, and individuals are ‘limited’ in their rationality or ability to make sense out of the variety of signals from the environment (March 1978). They emphasise that individuals frequently make choices based on incomplete information filtered through their subjective mental constructs, and simplify problems ‘because of the difficulties of anticipating or considering all alternatives and all information’ (March 1978:591-2).

In this context, ‘bounded rationality’ is one of the most widely known behavioural models describing human choice in complex situations. The model was originally proposed by economist and political scientist Herbert A. Simon (Simon 1955, 1956) and over the years it has been adapted by other scholars (March 1978, Jones 2003, Ostrom 2005). The model builds on social and cognitive psychology and organisational studies. ‘Bounded rationality’ draws on the assumption that intendedly rational behaviour is behaviour ‘within constraints’ produced by such aspects as limited information processing capacity, utilisation of memory and ability to learn (March 1978).

The ‘bounded rationality’ model, which is applicable to both individuals and organisations, is based on four major assumptions: first, people are goal oriented, but fail to accomplish goals due to cognitive limits; second, human thought is adaptive and decision-makers are able to improve their understanding with time; third, in the face of uncertainty humans face difficulties in calculating probabilities; and fourth, in evaluation of trade-offs humans choose alternatives that are satisfactory or ‘good enough’ (Simon 1955, Jones 2003). In other words, while institutional actors are assumed to act rationally and respond to institutions in the same way, ‘bounded rationality’ explains diverse behaviour by incorporating cognitive variables constraining the logic of action (Jones 2003).

Different models imply differing assumptions with regard to the role of institutions in shaping the ways the actors perceive the environment. Rational Choice scholars, in particular economists, tend to approach institutions as coordinating elements which enable human actors to deal with complexities and reduce information gathering or transaction costs (North 1990, Vatn 2005). The link between complexity and institutions was captured by the economist Ronald A. Heiner (Heiner 1983), who presented an argument that rule-following behaviour occurs in response to uncertainty and inability of agents to interpret environmental signals. As a result, agents are
willing to restrict flexibility to a limited set of choices to increase predictability of interactions (Heiner 1983).

In contrast, sociologists place emphasis on social determinants of human behaviour (Scott 2001). They argue that human ‘models’ of reality are social constructs, and understanding of the situation is gained through collectively produced concepts and shared discourses accumulated in institutions (Berger & Luckmann 1991). The way actors approach a decision-making situation is also determined by institutional or organisational roles which define norms and expectations linked to particular social positions (March & Olsen 2008).

Preferences or valuations ‘that institutional actors assign to actions and outcomes’ (Ostrom 2005:103) are the most debated component of human behaviour. Institutional theorists and analysts employ different assumptions regarding ‘rationality of preferences’ (Kato 1995). Building on the foundation of economic theory, Rational Choice analysts draw on the assumption that human actors interact with an intentional purpose to advance or maximise self-interest (North 1990, Ostrom 2005). Their core behavioural model is that of ‘economic man’ or ‘rational egoist’ (Ostrom 2005), whose actions are primarily driven by calculative utility- or wealth- maximisation strategies. The utility function, however, is not limited to monetary rewards, but may incorporate different material and non-material benefits that actors value in specific action situations (North 1990). For example, Public Choice Theory explains complex interactions in the political sector by employing such assumptions as material interests of voters, budget-maximising interests of bureaucracy, and re-election interests of politicians (Buchanan 1984). The so called ‘logic of collective action’ is seen as driven by preference aggregation, or the need of people to meet their shared interests (Olson 1971). Collective outcomes, therefore, tend to be examined against some normative standards of Pareto optimality, where improvement for the group is seen to be achieved if actors can improve their welfare without making anyone else worse off (Oppenheimer 2010).

Drawing on various theories stemming from sociology, anthropology, as well as psychology, Sociological Institutionalism offers a much more complex and diverse picture of human motivation. As members of social structures, both individuals and organisations are seen as being ‘socialised into patterns of thinking, roles and responsibilities’ (Vatn 2005:204). It is assumed that institutional actors act not just based on calculations of their personal benefits, but also on what they perceive to be right or appropriate in particular social contexts (Scott 2001, March & Olsen 2008). According to Scott (2001:68) the rationality of action is seen as shaped by the moral framework ‘that takes into account one’s relationships and obligations to others in the situation’.

Sociologists emphasise that collective choice decisions, or the ‘logic of social practice’, incorporate such aspects as social learning regarding the problem, and perception and preferences of other actors (Young 2002). Therefore, the socialisation process can transform the
ways in which involved actors reason about the problem, and produce solutions which cannot be predicted from the aggregation of individual preferences (Checkel 2005). This ‘logic’ has gained particular attention in the context of the Habermas (1984) theory of ‘communicative action’, which places emphasis on the role of language and communication as a basis for consensus and action (e.g. Healey 1997, Risse 2000).

The two institutionalist platforms also offer different perspectives on institutional mechanisms and their role in shaping the preferences of their actors. Rational Choice models tend to treat preferences of individual actors as ‘stand alone’ and exogenous to the decision-making situation (Young 2002). Consequently, institutions are viewed as sets of rules that constrain or enable particular choices, and provide for ‘checks and balances’ to produce the required equilibrium in the diversity of preferences (Shepsle 1989, North 1990). It is assumed that motivation to act and comply with rules is predominately affected by incentive and disincentive mechanisms which alter cost-benefit calculations (North 1990). These mechanisms are also regarded as the core determinants of the ability of large groups of actors to act in their common interest and to resolve social dilemmas (Olson 1971, Plott 1983). While some studies have revealed that norms of reciprocity, trust and reputation are equally important determinants, these findings are linked to interactions involving small groups of individuals sharing common rules and having the opportunity to communicate (Ostrom 2005).

Drawing on the assumption of values and preferences as social ‘constructs’, sociologists ascribe a much broader role for institutions (Young 2002). Institutions are seen not only as constraints but also as constitutive of their actors and interests. Therefore, rationality of preferences of institutional actors to pursue a particular course of action and comply with rules is explained through their involvement with institutions (Peters 1998, Scott 2001). As March and Olsen (2008:689) summarise:

Rules are followed because they are seen as natural, rightful, expected, and legitimate. Actors seek to fulfil the obligations encapsulated in a role, an identity, a membership in a political community or group, and the ethos, practices and expectations of its institutions. Embedded in a social collectivity, they do what they see as appropriate for themselves in a specific type of situation.

Internalisation of norms can occur through passive acceptance of institutional roles, or active and reflective internalisation of a new understanding of appropriateness as the ‘right thing to do’ (Checkel 2005). Furthermore, some of these roles can become internalised via the process of ‘habituation’ and expressed as habits or routines (Hodgson 2006, Fleetwood 2008).

While the literature tends to discuss different perspectives on institution-actor relations by contrasting Rational Choice and Sociological platforms, they are not seen as contesting or incompatible. Behavioural theories of both individual and collective choices are continuously evolving, and analysts from both directions agree that there is no single model that can be applied in any given decision-making setting (Scharpf 1997, Scott 2001, Ostrom 2005, 2011, March & Olsen 2008). As March and Olsen (2008:702) argue:
Specific logics, such as following rules of appropriateness and calculating individual expected utility, can be good approximations under specific conditions. It is difficult to deny the importance of each of them (and others) and inadequate to rely exclusively on one of them.

Furthermore, representatives from both directions continue to explore opportunities to reconcile different rationalities to improve the explanation and prediction of human behaviour in specific institutional settings (Heikkila & Isett 2004, Checkel 2005, Ostrom 2005, March & Olsen 2008). It is acknowledged that the most challenging problem for institutional analysis is to reconnect different drivers of human choices, and to match the assumptions ‘to the structure of the situation or linked set of relevant situations’ (Ostrom 2005:117).

In sum, as this brief review suggests, different interpretations of human behaviour have been employed as foundations for institutional studies. They are built on different assumptions regarding institutions as causal mechanisms affecting the ways in which human actors perceive decision-making situations and make their choices. How to approach this diversity of perspectives is addressed next.

2.2.3 Institutional functions

The diversity of ‘rationality’ assumptions applied by various disciplines of the social sciences has led to the emphasis on different institutional mechanisms being seen as the core determinants of human actions and interactions. Several attempts have been made to create some order in the myriad of theories, concepts and models by distinguishing various institutionalist directions (Hall & Taylor 1996, Goodin 1998b, Peters 1998) or institutional ‘pillars’ (Scott 2001). However, a common classification of institutional functions is still lacking. The core aim of this section is to ‘fill this gap’ by linking the diverse perspectives on institutional determinants of human choices in order to create a common platform for the analysis.

This study builds on the argument that different theories on institutional determinants of human choices offer complementary rather than conflicting insights into ‘how institutions work’. Therefore, to address the question of institutional causality, it proposes a new typology of institutional functions, which captures the diversity of institutional effects while differentiating among the various theoretical perspectives outlined previously. The typology builds on different ways in which rules can influence the three behavioural elements: perception, preferences and capabilities. As a result, this study identifies eight generic institutional functions: coordinative, cognitive, constitutive, normative, payoff, constraining-enabling, allocating and structuring2.

These functions are briefly discussed below.

The coordinative function is probably the most common and least debated institutional function. Institutions evolve to reduce the uncertainties which arise from the limited abilities of human actors to capture decision-making situations. Consequently, coordinating rules prescribing ‘in

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2 Note: the names of institutional functions do not imply a reference to particular theoretical or philosophical concepts such as ‘constitutive rules’ employed by Searle (1995) or ‘structuration’ introduced by Giddens (1984).
situation x do y’ form an integral part of any social organisation involving a large number of actors. Expressed in the form of standard operating procedures, instructions, guidelines, recommendations and other ‘how to’ prescriptions, they frequently become the core determinants of human actions. These rules are observed, not because they are considered as appropriate, benefiting or most effective, but because the actors simply ‘cannot conceive any other way of acting’ (Scott 2008:429). Coordinating rules are also the crucial element of many organisations requiring a high level of coordination (e.g. production lines), or in emergency and high risk operations (e.g. emergency services, traffic organisation, army operations). In other words, while there is a large diversity of potential choices that could be made, the existence of institutions limiting these choices is the crucial condition that enables coordinated and predictable actions of ‘boundedly rational’ individuals and organisations.

The cognitive function relates to the rules affecting the ways in which human actors interpret decision-making situations. This function is particularly distinguished by sociologists who emphasise the role of language and common discourses which shape the ways actors perceive or interpret particular problems (Berger & Luckmann 1991, Powell & DiMaggio 1991). Institutions, as shared and commonly accepted collections of conceptions of ‘environmental phenomena’ (Berger & Luckmann 1991), enable the actors to interpret a decision-making situation or establish ‘what is’ (Schmidt 2008). They can determine what information or data is relevant, where to look for precedents, how to interpret ambiguous provisions, and which interpretation is authoritative (March & Olsen 2008). Thus, as Ingram and Clay (2000:539) summarise, they provide ‘interpretative lenses for social facts, including other institutions’.

Moreover, institutions also operate as sources or transmitters of new knowledge and concepts unfamiliar to the wider society. Institutions can introduce new vocabularies and formalise many concepts of technological, philosophical or scientific origin (Stinchcombe 2001). For example, such concepts as ‘biodiversity’ and ‘sustainable development’ have been introduced to the global society through international agreements. These concepts have produced fundamental shifts in the ways human actors reason about environmental problems.

The constitutive function describes institutional effects on the perceived status of participating actors. In practice, what organisations can be created, for what purposes, and what roles individual actors can play in these organisations and in other social settings, is determined by the institutional framework (North 1990, March & Olsen 2008). In contrast to the cognitive function, which shapes the understanding of ‘what is’, constitutive rules enable human actors to classify themselves and others, or to establish ‘what am I’ and ‘who are they’ (March & Olsen 2008) in a specific situation. Drawing on Searle (1995) these rules can be described by using the syntax ‘x counts as y in context c’. Constitutive rules specify the conditions under which individual and collective actors can obtain or are obliged to hold particular social positions or roles, and the forms of behaviour that these roles incorporate. According to Simon (1991:126) the role can also be described as ‘a system of prescribed decision-premises’ which tells
‘organisation members how to reason about the problems and decisions that face them: where to look for appropriate and legitimate informational premises and goal (evaluative) premises, and what techniques to use in processing these premises.’ Shared understanding of established roles produces reciprocal expectation of particular behaviour (Simon 1991).

The normative function is performed by rules affecting the ways in which institutional actors evaluate their choices in specific situations. According to Ostrom (2005:112) norms can be described ‘as shared concepts of what must, must not, or may be appropriate actions or outcomes in particular types of situations’. Normative rules, thus, ‘introduce a prescriptive, evaluative and obligatory dimension into social life’ (Scott 2008:428). They affect the choices of institutional actors by providing guidance regarding what actions are appropriate and in what contexts. Behavioural response can occur as a result of internalisation of norms as beliefs that particular behaviour or choice is the ‘right thing to do’, or as a result of social pressure (Dequech 2006). Normative function, however, is not the exclusive domain of informal institutions. Constitutions, statutes, codes of conduct and other formal systems incorporate norms specifying the range of appropriate behaviour and aiding interpretation of other rules. In practice, both types of rules are interrelated, as informal social norms create the foundation for statutory systems, shaping priorities and defining ‘appropriate’ ways of regulation (Stinchcombe 2001, Scott 2008).

The payoff function is another institutional function that shapes the preferences of individual and collective actors. In contrast to norms, payoff rules are designed to affect choices by altering ‘wealth maximisation’ or ‘utility’ calculations. Payoff rules are the analytical domain of Rational Choice scholars, in particular economists, who focus on rewards-sanctions elements as core determinants of valuation and preferences (North 1990). This function, however, is not confined to market institutions. Taking into account the diversity of benefits or ‘utilities’, payoff rules can be expressed in numerous ways. For example, non-compliant behaviour can produce monetary (e.g. fines), as well as non-monetary (e.g. imprisonment) penalties. Similarly, institutions can reinforce desired behavioural outcomes by providing monetary (e.g. tax allowances, pay-rise) or non-monetary (e.g. entitlement to promotion, re-election) rewards. Thus, payoff rules alter the set of choices that could be made in a particular situation by introducing further consequences tied to specific actions or outcomes (Ostrom 2005).

The enabling-constraining function is predominately linked to capabilities. According to Scharpf (1997:43) ‘capabilities’, defined as ‘all action resources that allow an actor to influence an outcome in certain respects and to a certain degree’, need to be considered as a separate behavioural element. While some of the ‘action resources’ such as physical characteristics, intellectual abilities, education or experience, can be described as ‘personal’ properties (see Scharpf 1997), most of them are institutionally shaped. Rules, both formal and informal, enable or constrain actions by defining the rights and obligations and setting boundaries for the authority domain of particular positions. Usually expressed as ‘x is allowed, required or prohibited to do y in situation c’ these rules establish the sets of potential choices individuals and
collective actors can make. On the governance scale, these rules determine the level of power held by separate actors in such aspects as definition, change or enforcement of rules, and resource allocation or distribution. Enabling-constraining rules may or may not be supported by payoff rules specifying sanctions for the violation of the authority boundaries.

The *allocating* function is another determinant of capabilities. It can be argued that any behavioural model is based on the explicit or implicit assumption that institutional actors evaluate choices, and select actions in the light of the constraints imposed by available resources. Allocating rules operate as important determinants of actors’ abilities to use their rights or to take up particular positions. While financial and other resources tend to be addressed as ‘personal’ property of individuals and organisations (Scharpf 1997), more often than not they are institutionally formed. In practice, the functioning of current industrial societies is increasingly dependent on systems of rules enabling accumulation and (re)distribution of resources. For example, at the individual level, such instruments as taxes, fees or subsidies can produce shifts in the ‘affordable’ scope of actions and affect valuation and preferences. On the governance scale, the potential scope of functions or services provided by governmental agencies or local governments strongly depends on established budgets. Furthermore, allocating rules may become important drivers of power shifts and changes in relations between the actors.

Finally, the *structuring* (framework) rules shape so called ‘collective’ capabilities of groups of actors to interact and achieve shared goals. While individual choices are important factors in numerous situations, many significant decisions in current social systems are the product of interactions between few or many actors. In practice, all ongoing interactive processes are enabled and shaped by established ‘rules of the game’, which regulate access and modes of interaction (Scharpf 2001). They can range from contracts concluded between two individuals or corporate organisations, up to complex structures determining the operation of courts or policy-making procedures at state, national and international levels. Unlike ‘coordinating’ rules, they do not prescribe specific behaviour, choices or the result ‘of the game’ and, therefore, their influence on behavioural outcomes is less predictable. At the same time, in the political arena some structuring rules such as decision rules (e.g. majority rule, consensus, and veto rights) can alter the outcomes by determining the level of influence each actor has over the final decision (Shepsle 1989).

In summary, as the above typology reveals, there is no such a thing as an ‘institutionally free world’. Institutions, both formal and informal, can be regarded as performing a broad range of functions, and varying in their influence on behavioural outcomes. All functions are interrelated, and human choices and interactions are the products of ‘aggregated’ effects. Therefore, which functions and respective institutionalist theories, models and criteria are relevant, and which can be ignored or excluded from the analysis of causal influence, depends on the specific circumstances of each particular case. Thus, an important task of the analyst is to ascertain which functions ‘are at play’ in a given context (Scott 2008:429).
2.3 Institutional performance and effectiveness assessment: conceptual framework

The first part of this chapter clarified two core conceptual problems, namely, what is meant by the term ‘institutions’ and how institutions influence, or are assumed to influence, human actions and interactions. Building on this foundation this section has two major objectives. Firstly, to clarify and define what is understood by institutional effectiveness and what elements form the assessment structure. Secondly, to discuss the conceptualisation of institutional environmental performance and its implications for assessment design.

This part was considerably assisted by studies conducted on international environmental regimes which formed part of the Institutional Dimensions of Global Environmental Change (IDGEC) project (Zürn 1998, Miles et al. 2002, Young 2002, Underdal & Young 2004, Young et al. 2008, Mitchell 2008, Breitmeier et al. 2011). Building on this intellectual work, this section offers further clarification of the identified conceptual and analytical problems, and discusses them in the context of institutions operating at various levels of social organisation, with a particular focus on statutory systems.

2.3.1 Institutional performance and assessment design

‘Effectiveness’ is a widely employed concept in evaluation (Vedung 2010). In general, the evaluation literature shares some common understanding of ‘institutional effectiveness’ following its ordinary meaning as ‘the degree to which something is successful in producing a desired result’ (The Oxford Dictionary). For example, Young (2002:55) defines the effectiveness of environmental regimes broadly as ‘the capacity of these arrangements to prevent undesirable environmental changes and to solve environmental problems once they arise’. Underdal (2002:11) describes regime effectiveness as ‘the extent [to which] it successfully performs a certain function or solves the problem that motivated its establishment’. Mickwitz (2003:425) links effectiveness of environmental policy instruments to ‘the anticipated effects in the target area in relation to the stated objectives’.

While such generic definitions tend to place emphasis on ‘desired result’ or ‘success factor’, they do not reveal how institutions perform. As already discussed in the first part of this chapter, institutions both formal and informal are social constructs and as such cannot ‘resolve problems’ directly. As North (1990:107) has put it: ‘We cannot see, feel, touch, or even measure institutions; they are constructs of human mind.’ As ‘systems of rules’, institutions can achieve their goals only through the behavioural response of targeted actors. Therefore, this study proposes to define:

‘institutional performance’ as an institutional influence on, or contribution to, the behavioural response of targeted actors; and
‘institutional effectiveness’ as the extent to which an institution influences, or can potentially influence, the behavioural response of targeted actors towards the outcomes an institution has been designed to produce.

Linking institutional effectiveness to the achievement of institutional goals, however, is not sufficient for assessment design. As Underdal (1992, 2002) points out such definition lacks the precision to be used as an analytical tool. He argues that the design of the conceptual framework has to address at least three questions: ‘(1) what precisely constitutes the object to be evaluated? (2) against which standard is this object to be evaluated? (3) how do we go about comparing the object to this standard?’ (Underdal 2002:5). In other words, effectiveness assessment requires distinguishing the core components of the assessment structure. Definition of these components, available options for the assessment design and related conceptual and methodological challenges are sequentially discussed below.

**Institutional effects**

In general, effectiveness assessment builds on the assumption that any institution, organisation, program or policy is created for a particular purpose and aims to produce certain effects. As Helm and Sprintz (2000:633) emphasise the identification of effects that can be caused by the regime (i.e. institution) ‘constitutes the very essence of research on regime effectiveness.’ In practice, the designers of many formal institutions, such as statutory systems or international agreements, tend to incorporate clauses stating for what purpose a particular arrangement has been established and what goals and objectives it aims to achieve. Furthermore, the legal discipline offers a broad set of guidelines supporting consistent interpretation of rules and behavioural outcomes in the light of prescribed or implied institutional goals (Posner 1983, Eskridge & Frickey 1990). Interpretation of such goals for assessment design, however, encounters several conceptual and analytical challenges.

In practice, the availability of goal qualities facilitating effectiveness assessment in the managerial sense (i.e. measurable, location and time-bound) in institutional designs is rather an exception than a norm. As the analytical literature emphasises, institutional goals and objectives may be vaguely formulated, unrealistic or even contradictory (Bernauer 1995, Young 2002, Vedung 2006). Institutions may contain a ‘goal catalogue’ including vague requirements to balance competing or conflicting goals (Vedung 2006). In other cases the goals of interest may not be ‘stand alone’ statements, and the assessment may first require ‘disaggregation’ of goals into ‘several dimensions of variation’ (Bernauer 1995:367). Identification of ‘anticipated effects’ may also require interpretation of the content of rules. For example, as Mitchell (2008) notes, institutions may target behaviours which affect environmental quality without providing definition of such goals. In such cases the analyst cannot rely on institutional definitions, but must elicit embedded environmental goals (Mitchell 2008). Furthermore, the goals may not be determinate or static. Some problems may require periodic adjustments in institutional
arrangements to adapt to changing circumstances, or may result in new insights leading to problem reframing (Young 2004). In other words, detailed specification of what effects an institution undertakes to achieve, may involve a substantial level of interpretation and judgement.

Apart from interpretation, another question revolves around which ‘desired effects’ should be included in the assessment scope. In the evaluation literature there is a growing emphasis on multi-dimensionality of institutional performance and produced effects (Bellamy et al. 2001, Mitchell 2008, Breitmeier et al. 2011). In practice, achievement of any overarching institutional goal can be linked to a variety of configurations of rules pursuing specific (i.e. subsidiary) goals and objectives. For example, as Mitchell (2008) suggests, evaluation of international environmental regimes may require consideration of such effects as changes in public commitments and policy outputs, improved scientific understanding of a problem, creation of environmental norms, as well as distribution of economic costs and benefits. Institutional designs may also be required to meet certain ‘good governance’ standards such as transparency, legitimacy and accountability (Mitchell 2008). Therefore, the assessment of complex institutional systems often implies determining the boundaries of the effects under consideration. Inevitably, the inclusion or exclusion of particular sub- or side-effects can affect evaluation outcomes.

The diversity of institutional effects raises another ‘boundary’ question: ‘how wide should we cast the net in the search for impacts or consequences?’ (Young & Levy 1999:10). Institutions have a potential to produce a large range of effects: intended and unintended, direct and indirect, as well as positive and negative (Young & Levy 1999, Vedung 2006). Currently, there is no unified approach to defining which types of effects should form part of the effectiveness assessment. The IDGEC research literature (see Underdal & Young 2004) proposes to draw a distinction between simple effectiveness assessment and the assessment of broader consequences. According to Young (2004:5) simple effectiveness assessment considers the direct or indirect effects that institutions produce ‘within their own behavioural complexes or issue areas’. In contrast, consequence assessment involves the analysis of a range of indirect and mostly unintended and unforeseen effects that institutions may produce beyond their scope of regulation (Young 2004). For example, institutions may affect behavioural outcomes or biophysical impacts regulated by other institutions (‘cross-institutional effects’), or in combination with other institutional and non-institutional factors contribute to aggregated effects (‘systemic consequences’) (Young 2004).

Drawing boundaries between effects and consequences, however, is problematic for the analysis of ‘lower level’ systems. Institutional systems operating at the state level are interlinked and need to be viewed in configuration (McGrath 2010). Such a complex problem as biodiversity loss is inevitably a ‘cross-institutional effect’. Attempts to identify which effects were ‘intended’ and which were ‘unintended’ in the design of a particular sub-system would highly complicate
the analysis. Therefore, for the purposes of this study the concept of institutional ‘effects’ is used in a broader sense to describe any behavioural change that can, either fully or partially, be attributed to a particular institution.

Finally, interpretation of institutional effects raises another question: should ‘effectiveness’ be approached only in ‘positive’ terms? In the evaluation literature the concept of program or policy effectiveness tends to be linked to positive or beneficial contributions (Rossi et al. 2004, Vedung 2006). As Vedung (2006:407) points out intended effects ‘are by definition anticipated as well as positively valued by the intervention adopters.’ This study, however, supports the argument that the concept of ‘institutional effects’ should be treated as value neutral. There is no objective measure of what can be considered as ‘positive’ or ‘negative’. Institutions have been designed for different purposes. For example, an institution enabling behaviour which produces adverse environmental effects can be equally evaluated in positive terms of ‘contribution to the economic growth’ and in negative terms of ‘continuous biodiversity destruction’. Furthermore, as Young (2004) points out participants of the same regime may not share the same understanding of the problem and may hold divergent opinions on which results or effects are desired. At the same time, the terms ‘positive’ and ‘negative’ can be applied to describe those institutional effects which either contribute or create barriers to the resolution of a particular problem (see Young & Levy 1999).

**Assessment object: performance dimensions and criteria**

The evaluation of institutional performance highlights another question: what exactly is evaluated or what is the assessment object? In the evaluation literature, definition of the assessment object is often assisted by the systems approach introduced to policy analysis by Easton (1965). In the context of environmental regimes, the IDGEC research literature distinguishes between three core elements describing institutional performance: institutional ‘outputs’, ‘outcomes’ and ‘impacts’ (see Miles et al. 2002). The term ‘output’ is applied to describe institutions as an ‘end’ product of the policy formation process (e.g. international agreement, statute, regulation or plan) (Underdal 2002). ‘Outcomes’ describe changes in human behaviour resulting from the introduction of rules (Underdal 2002). Finally, ‘impacts’ describe ‘consequences that materialise as changes in the state of the biophysical environment itself’ (Underdal 2002:6). Thus, ‘outputs’, ‘outcomes’ and ‘impacts’ are linked as ‘three distinctive steps in a causal chain of events, where one serves as a starting point for analysing the subsequent stage(s)’ (Underdal 2002:6). Causal relations between ‘outputs’, ‘outcomes’ and ‘impacts’ are outlined in Figure 1.
The distinction between the three elements allows clarification of two core parts of the assessment object, namely ‘performance dimension’ and ‘performance criteria’. Institutional systems are complex. In practice, selection of a set of institutional effects for the analysis implies selection of particular configurations of rules (‘performance dimensions’) operating as the core determinants of respective behavioural response. Analysts from different disciplines such as economics, sociology and policy science are interested in different aspects of institutional performance and, therefore, place emphasis on different performance dimensions. For example, effects or impacts such as decreased or increased pollution levels could be traced back to the rules sanctioning polluters or providing incentives to improve production technology. Alternatively, produced agreement on the ‘acceptable’ levels of pollutants (outputs) could be linked to the decision-rules structuring the policy-making process. Thus, which performance dimension(s) form part of the assessment object depends on both the selected scope of effects and established causal linkages.

Conceptually, this study proposes to distinguish ‘performance dimensions’ from value criteria against which institutional performance can be evaluated. ‘Performance dimensions’ are configurations of rules contributing to the behavioural response of targeted actors which produces particular effects. ‘Performance criteria’ are normative statements specifying what aspects (effects) of institutional performance are evaluated.

The diversity of functions most institutions can perform (see section 2.2.3) and the diversity of produced effects imply a vast range of applicable criteria. Several attempts have been made to系统atize criteria into thematic groups (Mickwitz 2003, Vedung 2006, Mitchell 2008). However, a common typology is lacking. Therefore, to identify different options for assessment design this study distinguishes between two broad approaches to the selection of assessment

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3 Note: in the evaluation literature these terms tend to be applied interchangeably (see Mitchell 2008)
criteria. Drawing on the ‘logic of causality’ (see Figure 1 above) they can be described as: behavioural response-based and performance dimension-based.

Behavioural response-based criteria form an integral part of evaluative studies. In essence, effectiveness assessment seeks to answer the question about the extent of produced behavioural responses (i.e. outcomes), which is the core evidence of institutional performance. Based on the type of evidence indicating behavioural response, performance criteria can be divided into three major groups: output-based, outcome-based and impact-based. In other words, the extent of behavioural change can be evaluated using tangible outputs such as designed protocols, plans, policies or monitoring programs (i.e. output-based criteria). Alternatively, the assessment can focus on observed changes in the behaviour of targeted actors. This can be indicated, for example, by improved resource management practices, changes in decision-making patterns or attitudes (i.e. outcome-based criteria). Finally, institutional performance can be linked to the changes in the biophysical environment. In the environmental performance context such impact-based criteria as restored or cleared vegetated areas, increased or decreased environmental pollution levels, or changes in the resource abundance can be of particular importance. Challenges in the selection of behavioural response-based criteria and related implications for the assessment of institutional environmental performance will be discussed in more detail in the final part of this chapter.

Performance dimension- or rules- based assessment can be broadly described as the assessment of institutional potential to produce effects without evaluating the extent of achieved ‘on ground’ behavioural or biophysical change. This approach to effectiveness assessment is questioned in the evaluation literature (Vedung 2009). However, as will be discussed in the final part of this chapter, it may be required for the evaluation of institutional environmental performance. Performance dimension- based criteria can be grouped based on different perspectives taken on the evaluation. Drawing on the work of other analysts (Sabatier & Mazmanian 1980, Bellamy et al. 2001, Rossi et al. 2004), this study distinguishes between ‘contextual’, ‘systemic’, ‘causal theory’ and ‘financial’ groups.

The ‘contextual’ perspective seeks to address the problem of so called ‘institution - context fit’. A range of external factors can influence the potential of designed systems of rules to produce required behavioural responses. Therefore, the extent to which they ‘fit’ into the respective socio-cultural, economic and political frameworks is regarded as an important effectiveness determinant (Goodin 1998a). ‘Systemic’ evaluation approaches institutional effectiveness from the perspective of how well the rules ‘fit’ within a particular institutional system. With formal institutions this tends to be the domain of the legal discipline which is concerned with matters of clarity and consistency of rules, their interpretation, compliance and legal enforcement. The

4 Note: Vedung (2009) argues against prospective studies and supports ‘result-based’ approach.
‘causal theory’ or ‘instrumental assumptions’ (Bellamy et al. 2001) approach involves evaluation of rules in terms of validity of the assumptions underlying their design. In practice, any system of rules is constructed based on a set of assumptions and expectation about how the problem could be resolved. Consequently, the analysis may require evaluation of applied behavioural models (see section 2.2.2), adequacy of technical or scientific theories about the nature of the problem or its causal factors, or different assumptions underpinning the implementation process (Sabatier & Mazmanian 1980, Bellamy et al. 2001, Rossi et al. 2004). Finally, the ‘financial’ perspective addresses different implementation aspects expressed in monetary terms. It largely seeks to answer the question about the extent to which an institution can produce a required behavioural response in terms of available or allocated financial resources, or the resource capacities of involved actors (see Sabatier & Mazmanian 1980). In this context the effectiveness may need to be measured in such terms as cost-effectiveness or cost-efficiency (Rossi et al. 2004, Mitchell 2008).

**Performance standards and measurement**

‘Performance criteria’ can be further distinguished from ‘performance standards’ which are other elements to consider in the assessment design. ‘Performance standards’ can be defined as normative points to which observed results can be compared to assess the magnitude of institutional influence (Mitchell 2008) or be classed as success, failure or satisfactory performance (Vedung 2006). In other words, performance standards are established to estimate the extent of institutional progress or the level of performance on selected criteria. According to Underdal (2002) defining an evaluation standard involves two steps: the first is to determine the point of reference against which institutional performance can be compared, and the second is to select a metric of measurement.

In general, ‘any attempt at measuring effectiveness will have to refer to the state of affairs at one particular point in time’ (Underdal 2002:13). In the assessment of formal environmental regimes the IDGEC literature distinguishes between two major options for the selection of reference points referred to as ‘actual-versus-counterfactual’ and ‘actual-versus-aspirational’ (Underdal 2002, Breitmeier et al. 2006). ‘Actual-versus-counterfactual’ assessment design compares institutional progress over time and attempts to establish whether an institution has produced an effect (Mitchell 2008). In this case the progress can be evaluated by selecting between two points of reference: ‘no-institution condition’ and condition under previous configurations of rules (Underdal 2002). The ‘no-institution condition’ measures institutional effectiveness in absolute terms, whereas comparison to the state of affairs under previous rules describes relative change (Underdal 2002). ‘Actual-versus-aspirational’ design evaluates institutional progress towards the achievement of a desired state of affairs (Mitchell 2008).

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5 Note: cost-effectiveness and cost efficiency criteria can be applied both to evaluate the potential of particular regulatory solution to produce required behavioural change, as well as to evaluate produced results.
The IDGEC research literature distinguishes between three types of ‘aspirational’ standards: goal attainment, problem solving and collective optima (Helm and Sprintz 2000, Underdal 2002, Breitmeier et al. 2006). ‘Goal attainment’ focuses on the assessment of progress towards the achievement of institutionally established performance standards, or it evaluates institutions ‘in their own terms’ (Mitchell 2008:88). The ‘problem solving’ approach involves defining standards based on the characteristics of the problem underpinning institutional design (Mitchell 2008). Finally, the ‘collective optima’ approach involves the assessment of institutional progress towards some concept of a ‘good’ or ‘ideal’ solution (Underdal 2002). Potential options for selecting performance standards are summarised in Figure 2.

**Figure 2 Points of reference and performance standards**

Source: author based on Underdal (2002) and Mitchell (2008)

In general, all points of reference can be applied in the assessment design and are complementary (Underdal 2002). Each of them, however, presents different analytical challenges. The assessment of absolute or relative effectiveness allows describing how well an institution has performed compared to the state of affairs under conditions of no regulation or previous configurations of rules (Underdal 2002). However, with large scale institutions, experimental or quasi-experimental assessment designs are hardly possible because empirical data on baseline conditions may not be available (Bennear & Colliagnese 2005). Therefore, the assessment may require the application of hypothetical assumptions which can be subject to the assessor’s bias (Underdal 2002). The assessment of institutional progress towards specified goals (i.e. ‘goal attainment’) may encounter fewer problems with data, as monitoring and data gathering may be prescribed (Mitchell 2008). However, problems may emerge in cases where institutional standards are vaguely formulated, ambiguous, flawed or fail to reflect the management problem (Vedung 2006, Mitchell 2008). ‘Problem solving’ and ‘ideal solution’ options enable the analysts to define their own standards. However, problems may emerge with determining what levels of performance would lead to ‘problem resolution’ or provide an ‘ideal solution’. In the absence of common agreement, such standards may be highly arbitrary or biased (Underdal 2002, Mitchell 2008). Furthermore, selection of external standards can be also affected by political considerations (Mickwitz 2003).
Finally, selection of criteria and standards for performance assessment requires decisions on the system of measurement. A large diversity of applicable criteria and standards imply a large range of approaches to the measurement of institutional performance. In general, the evaluation of institutional performance on selected criteria can be based on qualitative and quantitative methods or a mix of both. There are different conceptual frameworks and methods developed to examine various performance criteria and standards. For example, evaluation of institutional performance in such terms as cost efficiency or cost effectiveness implies application of a particular methodological framework developed by the economics discipline. In contrast, criteria and standards describing changes in the biophysical environment would require evaluation based on concepts and methods of the environmental sciences. In other words, as Bellamy et al. (2001:413) have noted ‘there is a strong case for methodological pluralism.’

In this context, one of the emerging approaches to the assessment, which deserves separate attention, is performance scoring. Performance scoring relies on assigning numeric or nonnumeric values to separate performance variables on a given scale (Mitchell 2008). The approach offers two major benefits. First, it allows for more systematic and comprehensive evaluation of institutional performance along a large variety of criteria. Second, it offers a good basis for making comparisons across institutions (Mitchell 2008).

The scoring approach has been widely applied in many studies in such fields as business, education, psychology and medicine. However, it is relatively new in institutional analysis. So far, performance scoring has been applied in the evaluation of similar types of institutions such as international environmental regimes (Miles et al. 2002, Breitmeier et al. 2006) and local land use planning systems (Berke et al. 1996, Brody 2003).

However, as the IDGEC literature suggests, application of performance scoring in institutional analysis involves several challenges. The biggest challenge is the design of a common data protocol. Different data protocols have been made available for specific studies. At the same time, there are no commonly accepted dimensions, criteria and standards guiding the evaluation process (see Breitmeier et al. 2011 for a summary). Problems can also be encountered in the design of performance scales. While different scales can be employed (i.e. nominal, ordinal, interval, ratio) they must be acceptable in terms of construct validity, accuracy, reliability, appropriateness to respective performance criteria and standards, as well as applicability for comparative studies (see Mitchell 2008 for a discussion). How to construct such scales for the analysis of institutional environmental performance is still unresolved (Hovi et al. 2003, Breitmeier et al. 2006). Finally, scoring does not necessarily offer an ‘objective’ measure of institutional performance. As Underdal (2002:11) notes, all assessments of effectiveness will ‘inevitably involve some element of subjective judgement or inference on the part of the analyst’.
In summary, this section proposed the definition of two core analytical concepts underpinning institutional analysis, namely ‘institutional performance’ and ‘institutional effectiveness’ and provided the conceptual framework for the assessment design. The core elements of the conceptual framework and related options discussed in this sub-section are summarised in Figure 3. As it indicates, the design of institutional assessment is a complex process which requires making choices on a range of elements which, in turn, may significantly affect analytical outcomes. Selection of the elements for the analysis of institutional environmental performance and related implications for the assessment design are discussed in the next section.

Figure 3 Effectiveness assessment of institutional performance: elements of the assessment design

Source: author
2.3.2 Institutional environmental performance: conceptualisation and implications for the assessment design

By definition, ‘environmental’ institutions structure human interactions with the natural environment. Therefore, the ultimate interest in their evaluation lies in determining how and to what extent particular sets of rules contribute to changes (beneficial or adverse) in the state of the natural environment. However, the assessment of institutional ‘environmental’ performance encounters several challenges. The concluding part of this section undertakes to define institutional environmental performance and its effectiveness. Drawing on the conceptual framework, it also discusses several analytical problems encountered in the evaluation of institutional environmental performance and the resulting implications for assessment design.

**Institutional environmental performance**

Drawing on the concepts discussed in the first part of this section, two new concepts are introduced, which are defined as follows:

‘institutional environmental performance’ is an institutional influence on, or contribution to, the behavioural response of targeted actors, which either directly or indirectly produces, or has a potential to produce, environmental effects (impacts);

‘effectiveness of institutional environmental performance’ is the extent to which an institution influences, or can potentially influence, the behavioural response of targeted actors towards the achievement of the environmental effects an institution has been designed to produce.

The concept of ‘institutional environmental performance’ is proposed to distinguish between different dimensions of institutional performance. As indicated previously, institutions can produce a large variety of effects. Therefore, their performance can be evaluated against such criteria as transparent decision-making processes, just distribution of environmental resources, economic efficiency and cost effectiveness. While all of these criteria are important performance measures, it can be argued that they only characterise ‘social’ or ‘economic’ performance aspects. For example, they do not indicate whether, or to what extent, particular configurations of rules ‘make ecological sense’ (Galaz et al. 2008) or can resolve the environmental problem.

The evaluation of institutional environmental performance requires establishing two causal linkages: first, between particular configurations of rules and the behavioural response and second, between the behavioural response and existing or potential changes in the biophysical environment. As outlined previously, identification of the first link is essential for all institutional assessments. As Mitchell (2008:84) points out, ‘evidence that an institution did not change human behaviours undermines any claim of that institution’s influence on environmental
quality, even in the face of dramatic improvements.’ The second link requires examination of
behavioural outcomes and their causal role in the resolution or creation of environmental
problems. Similarly, a lack of sufficient causal links between the behavioural outcomes selected
for the assessment and existing or potential changes in the biophysical environment cannot
justify the claim of environmental performance assessment.

The proposed division, however, does not imply that economic or social dimensions are not
related to ‘on ground’ environmental impacts or that related criteria should be ignored. In
practice, the scope of rules characterising institutional environmental performance cannot be
narrowed down to rules prescribing, prohibiting or permitting actions that produce
environmental impacts. As will be discussed in the following chapters, in the management of
complex environmental problems, such as biodiversity loss, institutions perform a large range of
functions and targeted behavioural outcomes are shaped by numerous configurations of rules.
Therefore, which performance dimensions operate as core determinants of the environmental
outcomes is a question that needs to be addressed in a problem, institution and actor specific
context.

**Institutional environmental performance: assessment challenges**

Ideal assessment situations with clear, measurable goals, performance standards, available
monitoring systems and data are rare. In real life, the decisions regarding the structure and
elements of the assessment design are driven by numerous methodological problems. The
remaining part of this sub-section discusses two significant problems affecting the evaluation of
institutional environmental performance: availability of relevant criteria characterising the extent
of behavioural change, and the identification of causality links along an institution – behavioural
change – environmental change chain of events.

In environmental performance assessment the ultimate interest of analysts and policy-makers lies
in impact-based criteria, or criteria describing the change in the natural environment attributable
to the regulations (Underdal 2002). As Mitchell (2008) notes, an application of such criteria is
particularly useful in cases when environmental change is dominated by anthropogenic drivers.
However, impact-based evaluation design often is not possible. The first and most common
problem is data availability. As Koontz and Thomas (2006) summarise, three core challenges are
data gathering, long time horizons between the actions and actual change in the natural
environment, and design of research protocols accounting for multiple variables. Data gathering
problems may also emerge due to insufficient environmental knowledge and a lack of resources
to support data gathering and monitoring (Mitchell 2008). The second problem is causality.
Linking changes in the biophysical environment to the regulations requires accounting for other
causal variables, both anthropogenic and natural, influencing the scope of produced impacts
(Bernauer 1995, Mitchell 2008). Consequently, drawing causal inferences for institutions along
an institution - behavioural change - environmental change chain may involve significant
methodological challenges (see Bennear & Colliagnese 2005 for a review regarding pollution regulation).

As the IDGEC literature suggests the ‘second-best’ alternative is to use criteria characterising produced behavioural response (i.e. ‘outcome-based’). An application of such criteria may be the only feasible option where institutions regulate either a small fraction of behavioural change, or behaviour that is not directly linked to environmental impacts (Mitchell 2008). As changes in human behaviour are easier to observe and measure, outcome-based criteria offer benefits in terms of data availability. Moreover, consistently collected statistical data may be available as environmentally related behaviour tends to be monitored for economic or social reasons (Mitchell 2008). These criteria also have advantages in terms of causality, as they are closer in the causal chain and therefore there are fewer external factors that need to be taken into account when distinguishing institutional influence (Mitchell 2008, Young 2008).

Despite the benefits, application of outcome-based criteria in the assessment of institutional environmental performance can encounter several problems. First is relevance, as selected criteria describing behavioural change may fail to provide reliable progress measures for potential changes in the biophysical environment (Mitchell 2008). For example, an assessment against observed changes in resource extraction levels may fail to provide information on whether resource stocks are maintained at a sustainable level. Similarly, measuring institutional performance in terms such as change in attitudes, management practices or levels of collaboration may not provide sufficient indication whether particular environmental problems have been or can be resolved. Second, problems of determining the level of institutional influence still remain. In practice, there is a wide range of intervening factors that can influence behavioural responses or produce the same effects (Underdal 2008). For example, as Mickwitz (2003) points out, behavioural change contributing to the achievement of decreased pollution levels may occur due to factors such as changes in technology or changes in market demand.

The easiest sets of applicable criteria for assessment are those characterising behavioural change in terms of produced ‘outputs’. In practice, it is relatively easy to demonstrate the causal links between an institution and the regulations, plans, policies or programs adopted as part of the implementation process (Young 2008). Evaluation against output-based criteria can be of particular relevance in cases where the assessment focuses on performance dimensions structuring the policy-making process or so called ‘collective-choice’ level (Ostrom 2005). But, selection of outputs as measures for institutional environmental performance in most cases will not be sufficient to determine institutional effectiveness in problem solving terms. As Koontz and Thomas (2006) note, the fact that involved actors have produced a set of documents may not provide a good indication regarding the extent to which they can contribute to changes in human behaviour affecting the environment.
Finally, the assessment design exclusively based on behavioural response criteria excludes a significant number of institutions which have not produced ‘actual’ outcomes. It is acknowledged, that some formal institutions may lack so called ‘behavioural significance’ (Young 2002) or can be described as ‘rules on paper’ (Ostrom 2005). However, an intended behavioural response may not have occurred for several other reasons. First, institutions can be newly designed or amended and, therefore, actual change in behaviour cannot be fully determined (Underdal 2008). Second, rules could be in place for a long period of time, but may trigger a behavioural response only under specific conditions, such as extreme events (e.g. water distribution under prolonged drought conditions). Third, a behavioural response may not have occurred due to other external factors. For example, a response to changes in the resource extraction regulation may not yet be observed due to such factors as decreased market demand or changes in other incentive mechanisms. And finally, there may be flaws in institutional design which preclude the required behavioural change or produce wrong outcomes (i.e. ‘perverse effects’).

**Implications for the assessment design**

The above challenges are difficult to resolve. In some cases the problem can be addressed by the involvement of multiple analysts or experts, selection of multiple data sources, or application of a variety of analytical methods (Mickwitz 2003). At the same time, in many situations the analysts are left with configurations of rules ‘as they are’, and a set of explicit or implicit assumptions regarding how and why they should contribute to the resolution (or creation) of a particular environmental problem.

This suggests that many assessment situations will require performance dimension-based design. Such assessment will involve the identification of mechanisms through which a particular institution regulates the behaviour, and the evaluation of ‘whether or not these mechanisms are working’ (Zürn 1998:639), or can potentially work within a particular contextual setting. Consequently, it shifts the traditional formulation of the ‘effectiveness’ question from ‘to what extent the institution has contributed to the achievement of the outcomes?’ to the question ‘to what extent the institution can potentially contribute to the achievement of the outcomes?’ In other words, using Ostrom’s (2007:15182) question, the evaluation seeks to establish ‘what patterns of interactions and outcomes are likely to result from using a particular set of rules in a specific context’.

Performance dimension-based assessment design provides several benefits. Obviously, it avoids identified limitations inherent in the selection of behavioural-based criteria and standards. More importantly, it does not limit the evaluation to institutions producing observable behavioural response. It enables both retrospective, as well as prospective studies. As a result, it significantly
expands the scope of rules which can be selected for the effectiveness assessment. Furthermore, it can be argued that prospective evaluation studies identifying potential problems with institutional designs can become important preventive measures.

Performance dimension-based assessment design does not automatically resolve all problems with assessment criteria and standards. The search for institutional design features that contribute to the achievement of environmental outcomes forms an important part of institutional studies (Ostrom 1990, 2005, Breitmeier et al. 2011). However, as the literature suggests, identifying and linking design features to particular environmental outcomes is a complicated exercise (Bernauer 1995, Zürn 1998, Young 2002, 2008, Mitchell 2008). Evaluation of institutional designs requires an understanding of the nature of institutional influence, behaviour of actors, problem characteristics, overall institutional setting and contextual factors. As the assessment is based on a range of assumptions, the projections regarding potential outcomes and impacts can be subject to various errors. How to systematically employ this strategy, identify a variety of institutional performance dimensions, and link them to diverse determinants of environmental outcomes will form the core of the next two chapters of this study.

2.4 Summary and conclusions

This chapter aimed to establish the theoretical and analytical platform for the study. Driven by the questions - what is understood by institutions? and what role do they play in structuring human behaviour? - the first part examined a range of theoretical problems regarding the conceptualisation of institutions and institution-actor relations. The second part defined the concepts of institutional effectiveness, institutional performance and institutional environmental performance, as well as identified and discussed major elements of assessment design.

As ‘systems of rules’ institutions evolve to create greater regularity and predictability in human interactions. In the theoretical literature the concept of ‘institutions’ is applied to a large variety of rules, ranging from formal regulations to the rules expressed as common discourses and routine actions. This study distinguishes between two interacting types of rules: formal and informal. Formal institutions are described as systems of rules that are articulated in constitutive documents such as laws, regulations and other regulatory instruments. Informal institutions are described as ‘unwritten’ rules that are embedded in established practices, shared discourses, informal understandings, roles and routines. This study also distinguishes between ‘institutions’ and ‘organisations’ as institutionally created actors. Rules structuring the behaviour of individual actors within the boundaries of a single organisation have been referred to as organisational rules.

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6 Note: in the evaluation literature some authors argue against the expansion of the evaluation scope and prospective studies (see Vedung 2009).
New Institutionalism offers a broad theoretical umbrella for different disciplinary perspectives on the institutional determinants of human actions. Significant differences in the assumptions regarding the extent of institutional influence can be found in the work of scholars associated with Sociological and Rational Choice directions of New Institutionalism. There is no single behavioural model which can be applied across institutional studies. Furthermore, the representatives from different disciplines place emphasis on different institutional functions. In response, this study proposes a typology of eight generic institutional functions which capture the variety of institutional effects identified in the theoretical literature. As the typology reveals, institutional and organisational rules can perform a broad range of functions and can be important explanatory factors of a large range of behavioural phenomena.

Institutional performance assessment is an evolving field of analysis which lacks a common conceptualisation of institutional performance and effectiveness. As institutions are not actors in their own right this study defines ‘institutional performance’ as an institutional influence on, or contribution to, the behavioural response of targeted actors. The ‘effectiveness of institutional performance’ is defined as the extent to which an institution influences, or can potentially influence the behavioural response of targeted actors towards the outcomes an institution has been designed to produce.

Assessment design is a complex process which involves determining which effects an institution aims to produce, which configurations of rules contribute to the achievement of specified effects, and against which criteria and how this performance could be measured. To identify the assessment object this study proposes to distinguish between the configurations of rules (performance dimensions) and different groups of value criteria and standards that can be selected for the evaluation. This study also introduces the concept of institutional environmental performance, which places emphasis on linkages between established configurations of rules and ‘on ground’ environmental problems an institution has been designed to resolve, or which it may inadvertently create.

The evaluation of institutional environmental performance encounters several methodological challenges revolving around the availability of relevant criteria and standards and identification of institution - behavioural change - biophysical change causality. Consequently, this chapter argued that, in a considerable amount of cases, the evaluation of institutional environmental performance will require the analysis of institutional statements or configurations of rules. This approach implies identification of critical configurations of rules (performance dimensions) having the potential to resolve or create environmental problems in specific contextual settings. Which institutional functions are considered as being critical for the achievement of biodiversity protection outcomes, and which determinants of their effectiveness have been distinguished in the academic literature, will be explored in the next chapter.
Chapter 3 - Biodiversity Protection and Institutions

3.1 Introduction

The previous chapter defined the effectiveness of institutional environmental performance in terms of the extent to which an institution influences, or can potentially influence, the behavioural response of targeted actors towards the achievement of the environmental effects an institution has been designed to produce. In the context of biodiversity protection, which is the focus of this study, this definition leads to two further questions: what role do institutions play in structuring human interactions with biodiversity? and what performance dimensions and criteria describe the institutional potential to achieve biodiversity protection outcomes? The purpose of this chapter is to examine the scope and characteristics of the biodiversity protection problem.

Conceptualisation of biodiversity, its protection and exploitation are topics addressed by a large body of literature produced by scholars from both environmental and social sciences. Given the vast scope of the problem, this chapter does not attempt to provide an in-depth review of different attributes of the problem (see Millennium Ecosystem Assessment for a comprehensive review). Instead, maintaining an ‘institutional lens’, it focuses on various attributes that can have implications for institutional design and analysis. This chapter addresses an array of generic questions - what should be protected where, why, how and by whom - which require placing the biodiversity protection problem in science, management and governance contexts.

The chapter consists of three parts. The first (section 3.2) specifies what is understood by ‘biodiversity’ and ‘biodiversity protection’ and what role biodiversity plays in human well-being. The second (section 3.3) explores ecosystem management as the dominant paradigm framing natural resource management policies and approaches. In particular, it examines the structure and application of adaptive management, which is widely promoted as the core approach for the management of complex environmental problems. The third (section 3.4) reviews several trends in the design and conceptualisation of local environmental governance, with particular focus on the distribution of power and administrative responsibilities.

Each part follows two steps. First, it explores major characteristics of the problem as presented or debated by the respective cluster of the literature. Second, it undertakes to summarise or discuss the implications the problem attributes may have for various elements of institutional design, such as involved actors and their mandates, goals and objectives, decision-making scopes, scales and processes. This review forms the basis for the analytical framework described in chapter 4 and subsequently the analysis of the Queensland Planning System in the second part of the study.
3.2 Introduction to biodiversity: *what, where and why* to protect

As understood by the scientific community, ‘biodiversity’ describes the variety of life in all forms, levels and scales (Calicott et al. 1998). The encompassing nature of the concept, however, has caused a great deal of confusion regarding *what* should be conserved or protected, *where* and *why*. This section undertakes to clarify the scope and characteristics of the problem as presented in the environmental science literature.

3.2.1 Biodiversity protection: problem characteristics

Since its introduction, the concept of ‘biological diversity’ or ‘biodiversity’ has gained popularity among scientists, politicians and the wider public (Jeffries 2006). Initially, the understanding of the concept revolved around species richness, which directed management activities towards conservation of rare species or areas with high species richness (Poiani et al. 2000). Recent interpretations, however, view biodiversity more broadly as a variety of living organisms, ecological complexes where they occur, and the ways they interact with each other and the environment (Noss 1990, Redford & Richter 1999). Apart from the well-known definition of the Biodiversity Convention (see chapter 1), a large variety of definitions of the concept can be found in the literature (DeLong 1996, Baydak & Campa 1999). Furthermore, as Noss (1990:356) argues, a definition ‘that is altogether simple, comprehensive, and fully operational (i.e. responsive to real life management and regulatory questions) is unlikely to be found’. Therefore, instead of focusing on a precise definition of the concept, this section addresses a variety of elements characterising biodiversity.

To examine the scope of the concept this study uses the hierarchical approach proposed by Noss (1990), which describes ‘biodiversity’ as consisting of several interconnected levels of biological organisation: genes, species, populations, communities and ecosystems. The hierarchy also incorporates ‘landscape’ as a ‘human-dominated’ level (Forman 1995). Each level of organisation consists of three primary attributes or components: composition, structure and function. ‘Composition’ describes the variety and identity of elements in each level, such as genetic constitution of populations, the identity and relative abundances of species within a community, and types of communities distributed across a landscape (Noss & Cooperrider 1994). ‘Structure’ describes how the elements of the system are organised physically, or the ‘pattern of a system’ (Noss 1990:357). For example, it includes such elements as dispersion and vertical layering of plants, logs in the forest, or patchiness of vegetation (Noss & Cooperrider 1994). The concept of ‘function’ refers to ‘ecological and evolutionary processes, including gene flow, disturbances, and nutrient cycling’ (Noss 1990:357). All these attributes form part of a nested hierarchy of interconnected levels of organisation.

Three levels of biological organisation have been frequently distinguished in the literature on biodiversity management at larger spatial scales: species, ecosystems and landscapes. Defined as
‘a group of actually and potentially interbreeding individuals’ (Krohne 2001:7), species are mostly considered as the smallest management unit to pursue biodiversity protection goals (Dale et al. 2000). Apart from protection of rare or threatened species, the focus on the species level has been supported for several other reasons. First, taking into account the complexity of biodiversity, particular groups of species have been proposed as surrogates to meet conservation objectives (Mac Nally et al. 2002). Protection of so called ‘umbrella species’, which are either at the top of the food chains or have large habitat ranges, allows protection of a number of species with smaller habitat ranges (Franklin 1993). Focus on ‘indicator species’ allows monitoring of the status of larger functional groups, or the quality of the habitat (Dale et al. 2000). Protection of ‘keystone’ species, which have a disproportionally large impact on ecological processes, has been recognised as crucial for the conservation and preservation of particular communities or ecological systems (Power et al. 1996). Second, biodiversity protection can gain wider public support if it includes ‘iconic’ or ‘flagship’ species which are particularly valued by the community (Franklin 1993). Finally, some species do not occur in predictable fashion within certain ecosystems and therefore their protection needs an individual approach (Groves et al. 2002).

The exclusive focus on species, in particular on rare, threatened or endangered species, has been criticised as insufficient to achieve biodiversity protection (Franklin 1993, MEA 2005a). Therefore, the preferred unit for biodiversity management is an ecosystem, described as ‘a dynamic complex of plant, animal and microorganism communities and the nonliving environment interacting as a functional unit’ (Biodiversity Convention, Article 2). While the Biodiversity Convention provides only a broad distinction between terrestrial, marine and other aquatic ecosystems (Article 2), conservation biology offers a number of approaches to further classify and locate ecosystems (Pojar et al. 1987, Klijn & Haes 1994, Margules & Presley 2000). For example, according to the Interim Biogeographic Regionalization for Australia (IBRA) framework for conservation, the continent is divided into 89 geographically distinct bioregions based on climate, geology, landform, native vegetation and species information, which are further divided into 419 sub-regions and regional ecosystems (DSEWPC 2012).

Recognition of humans as part of ecosystems and the importance of altered land-cover patterns in sustaining biodiversity have led to increasing focus on landscape management. The concept of ‘landscape’ has a variety of definitions (Wu 2013). For example, according to the European Landscape Convention ‘landscape’ is defined as ‘an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors’ (Article 1). As ecological units, landscapes consist of patches, corridors and a matrix which is ‘a major determinant of functional flows and movements through the landscape and of changes in its pattern and process over time’ (Forman 1995:135).

The concept does not entail distinct spatial boundaries; the focus of landscape management is rather on spatial pattern. Size, shape and spatial relationships of land-cover types influence the

In general, biodiversity protection is associated with the ‘natural’ or ‘native’ variety of species in a particular area (Redford & Richter 1999). One of the reasons for this distinction is the recognition that introduced species, while increasing richness in a particular location, threaten overall diversity (MEA 2005a). Biodiversity protection, however, is not constrained to pristine areas. In this context, clarification is required between two terms often interchangeably applied in the literature: ‘biodiversity preservation’ and ‘conservation’. As Redford and Richter (1999) suggest, the term ‘preservation’ predominately refers to non-use goals which can be pursued in areas with no or little human intervention. Biodiversity ‘conservation’, however, implies ‘consumptive and nonconsumptive use without complete destruction/conversion’ (Redford & Richter 1999:1247). For example, the Australian National Biodiversity Strategy 2010-2030 (NRMMCC 2010:84) defines ‘conservation’ as ‘the protection, maintenance, management, sustainable use, restoration and improvement of the natural environment’. In other words, biodiversity protection covers a range of goals which can be pursued in various areas, ranging from pristine to semi-modified, modified and highly modified systems such as urban areas (see Savard et al. 2000, Pickett et al. 2001 for urban biodiversity).

Thus, from the science perspective, biodiversity can be characterised as a broad and complex management problem, which encompasses multiple interacting levels of biological organization, and which can be managed in different types of areas and at different spatial and temporal scales. There is no single level for management, and different theories, principles and management approaches may be required for specific interventions (Redford & Richter 1999).

### 3.2.2 Biodiversity and human well-being or why to protect

The achievement of biodiversity protection goals inevitably involves the problem of values, or the question why protection is necessary. In general, biodiversity protection is driven by two value paradigms. The non-utilitarian paradigm acknowledges the intrinsic value of natural systems as ‘value in and for itself’ (MEA 2005a:34). The utilitarian paradigm is based on the satisfaction of human preferences, derived either from direct use (use values) or from the existence (non-use or existence values) of natural systems (MEA 2005a). With accelerating pressures on biotic resources worldwide, the utilitarian paradigm is gaining increasing dominance at the policy-making level. Therefore, this section examines the concept of ‘ecosystem services’ that currently underpins the utilitarian approach (Gómez-Baggethun et al. 2010) and its potential to support the achievement of biodiversity protection outcomes.

Biodiversity and ecosystems are related concepts. According to the Millennium Ecosystem Assessment (MEA): ‘diversity is a structural feature of ecosystems, and the variability among
ecosystems is an element of biodiversity’ (MEA 2005a:29). As a property or structural feature, biodiversity has been considered as the core determinant of the capacity of ecosystems to adapt to changing conditions, and to respond to external disturbances (Walker 1995, Christensen et al. 1996, Carpenter & Gunderson 2001). As Niemelä (1999:128) summarises, ‘biodiversity is both the raw material providing the basis for adaptations of ecological systems to their environments, and a buffer that enhances the system’s resilience against disturbances.’

The view on ecosystems as service providers first emerged in the late 1970s (Haines-Young & Potschin 2010). Over the years, the concept of ‘services’ has attracted the attention of scholars as a frame which allows expressing ecological concerns in economic terms and emphasises societal dependence on natural systems (Perrings et al. 1992, Costanza et al. 1997, Daily 1997). In the policy-making arena the concept gained wider recognition with the endorsement of the principles of the ecosystem approach to support the implementation of the Biodiversity Convention (COP 2000 Decision V/6). In 2004, the Biodiversity Convention introduced the definition of biodiversity loss as: ‘the long-term or permanent qualitative or quantitative reduction in components of biodiversity and their potential to provide goods and services’ (COP 2004 Decision VII/30).

The crucial turn in the popularity of the concept, was the publication of the MEA. The MEA defined ‘ecosystem services’ as ‘benefits that ecosystems provide’, and linked them to different aspects of human well-being (MEA 2005a:27). As a result, the concept has gradually evolved from a ‘pedagogic tool’ to gain public support to the use of schemes supporting monetization and commodification of ecosystem functions (Gómez-Baggethun et al. 2010).

A range of studies has been conducted on the description and classification of ecosystem services (Costanza et al. 1997, de Groot et al. 2002, Wallace 2007). Perhaps the most widely used classification is that of the MEA, distinguishing four major types of services: provisioning, regulating, cultural and supporting. According to this classification provisioning services comprise those related to the production of materials for consumption (e.g. food, water, timber). Regulating services include such processes as erosion protection, climate regulation, water purification, pollination. Cultural services are related to the cultural, aesthetic and other non-consumptive needs, such as recreation, spiritual and cultural connection. Finally, supporting services include all those required for service generation, such as soil formation, photosynthesis and nutrient cycling (MEA 2005a). Recently, the application of the concept has also been extended to other levels of organisation such as landscapes (i.e. ‘landscape services’) (Termorshuizen & Opdam 2009, de Groot et al. 2010).

While biodiversity has been linked to the provision of ecosystem services, understanding of these linkages differ. One of the attempts to clarify causal relations is the conceptual framework proposed by Haines-Young and Potschin (2010), which as de Groot et al. (2010) argue, reflects the growing consensus in linking ecosystems to human well-being. The framework distinguishes
between four components: ‘biophysical structures and processes’, ‘functions’, ‘services’ and ‘benefits and values’. According to the framework, the presence of biophysical structures and processes generates functions which can be described as the ecosystems’ capacity or capability to provide goods or services. Services satisfy human needs and, therefore, need to be considered in the context of potential beneficiaries. Benefits (e.g. nutrition, pleasure) are gained by the actual use of services which in some cases can be valued in economic terms (Haines-Young & Potschin 2010, de Groot et al. 2010). Thus, biodiversity is not conceptualised as a service in itself, but as an ‘enabling’ or capacity building property of ecosystems.

In practice, the application of the ‘services’ concept to valuation of biodiversity encounters several problems, the main one being correlation. From the ecological perspective, biodiversity protection and enhancement of ecosystem services are not perfectly correlated goals (Chan et al. 2007). Several studies have been undertaken to understand spatial relationships between areas rich in biodiversity and areas required for the provisioning of several ecosystem services (Chan et al. 2006, Egoh et al. 2009, Nelson et al. 2009, Bai et al. 2011). They have produced mixed results. For example, Chan et al. (2006) found a low spatial correlation between the six ecosystem services of carbon storage, flood control, forage production, pollination, recreation and water provision, as well as low average correlation between biodiversity and those services. The highest positive correlations with biodiversity were identified in carbon storage, water provision and recreation services. In another modelling study, Nelson et al. (2009) found that the scenario having a high score for a variety of ecosystem services (carbon sequestration, water quality and soil conservation) also scored high for biodiversity.

In general, there is a negative correlation between biodiversity protection and commodity production values (Chan et al. 2006, Nelson et al. 2009). As de Groot et al. (2010) point out, the provisioning services, such as food and timber, cannot be provided by pristine ecosystems. Thus, the use of these services adversely affects biodiversity values. Similarly, some regulatory services, such as carbon sequestration, can be provided by modified systems and non-native vegetation (Chan et al. 2007, de Groot et al. 2010). Thus, the extent to which the maintenance of ecosystem services can contribute to biodiversity protection outcomes is determined by the type and diversity of services under consideration.

Another problem concerns the knowledge base. Identification of the whole range of ecosystem functions providing services is a highly complex task. Availability of scientific evidence, especially on the local scale, presents significant challenges in the application of the ‘services approach’ in policy design and implementation (Tomich et al. 2004). While research has been conducted on several ecosystem elements which play important functional roles in service provision, the whole scope of linkages is far from clear (Kremen 2005, de Bello et al. 2008). Furthermore, the complexity of the problem and gaps in the existing level of knowledge requires significant investments in data acquisition. As Kremen and Ostfield (2005:547) argue, to gain
sufficient understanding of the ecology of ecosystem services the research ‘may require an investment akin to that devoted to agriculture, medicine, space exploration, or defence.’

Finally, recognition of ecosystems as service providers does not necessarily resolve value conflicts. Ecosystem services may have different values for local, national or international communities (de Groot et al. 2010). Management sectors may be interested in a few services, which may conflict with values of the broader community (Turner et al. 2003). As the MEA acknowledges, in the presence of strong economic incentives management sectors, groups of individuals or local communities may derive larger benefits from services resulting in biodiversity modification or loss (MEA 2005a). Differences in valuation may also occur when weighing short term gains against long term benefits (de Groot et al. 2010). Furthermore, there are concerns that the focus on utilitarian values may produce shifts in the logic of conservation from ethical obligation to economic self-interest (Gómez-Baggethun et al. 2010).

### 3.2.3 Biodiversity, science and institutions

The concept of biodiversity has been formulated by the scientific community. Over the last three decades different disciplines of environmental sciences, such as ecology, conservation biology and landscape ecology, have proposed a large variety of so called ‘laws of nature’ which have the potential to create greater predictability in human interactions with natural systems. But application of proposed theories, concepts, principles and management approaches has been rarely discussed in the context of operating institutional systems. Furthermore, there is an increasing recognition of ‘science-policy’ and ‘science-management’ gaps which, as scholars acknowledge, diminish the role of the environmental sciences in the policy-making process (Wu & Hobbs 2002, Fazey et al. 2005, Knight et al. 2008). Despite this gap, several themes having implications for institutional design and analysis can be distilled from the literature.

From the regulatory perspective, the concept of biodiversity is very broad. Therefore, institutional designs need to specify the scope of actions and expected outcomes pursued under an overarching biodiversity protection goal (see Dawson 2002 for a detailed analysis). It can be argued that generic requirements to consider biodiversity protection without any detailed guidance for the operational level can be treated as a ‘rule on paper’ (Ostrom 2005) failing to provide predictable behavioural change. Furthermore, at the current state of knowledge, understanding biodiversity and predicting all effects of actions on its components are far beyond the capacities of any single institutional actor, whether individual or collective. Consequently, the regulation of management interventions needs to translate various theories, concepts and principles into more specific sets of prescriptions that can structure decision-making in particular settings. This also suggests that institutional designs may need periodical re-evaluation of scientific theories and assumptions underpinning regulated management interventions.
The diversity of levels of biological organisation and scales implies multiple performance criteria and standards. Science does not provide universal sets of indicators (Redford & Richter 1999). As Noss (1990:357) summarises:

The hierarchy of the concept suggests that biodiversity be monitored at multiple levels of organisation, and at multiple spatial and temporal scales. No single level of organisation (e.g. gene, population, community) is fundamental, and different levels of resolution are appropriate for different questions.

In other words, any performance criteria and standards incorporated in institutional design need to be developed in relation to specific management interventions, relevant levels of biological organisations and spatio-temporal scales. As the MEA emphasises, ‘any single indicator, such as species diversity, is generally a poor indicator for many aspects of biodiversity that may be of concern for policy-makers’ (MEA 2005a:2).

Biodiversity protection is a highly complex problem. Limited understanding of complex interactions between different components of the biophysical systems and knowledge availability are widely acknowledged barriers for policy-making (Sutherland et al. 2006, Hanssen et al. 2009, Morton et al. 2009, Termorshuizen & Opdam 2009). In this context, information delivered by the environmental sciences has been criticised for being contradictory. This has enabled policy-makers and stakeholders to hold divergent positions in the rule setting and implementation process (Rayner 2006). Insufficiency of knowledge and conflicting perspectives also imply that it may be hard or even impossible to estimate to what extent adopted solutions (e.g. corridor widths, buffer zones or resource extraction limits) can resolve or create problems in specific settings.

From the perspective of institutional design, a lack of knowledge may lead to two potential flaws in regulation. First, institutional performance may be affected by flawed assumptions on the extent of available scientific knowledge, or the capacity of sciences to provide the required information within decision-making timeframes. As Lawton (2007) notes, the policy can be ‘ahead of science’, setting ambitious biodiversity protection goals far beyond the scientific capacities to state how to achieve them in practice. Second, even where information can be obtained, its application may face capacity constraints. Evaluation of impacts on ecosystems or their components, and monitoring and development of the knowledge base requires significant information processing capacity, resources, skills and expertise. In this context, insufficient funding and human resources have been frequently acknowledged as the key problems limiting the abilities of responsible authorities to acquire and apply environmental information (Szaro et al. 1998a, Stokes et al. 2009, Keene & Pullin 2011). To this end, institutional analysis may require reconsideration of the underpinning assumptions regarding the capabilities of both knowledge providers and users.

Finally, a frequently emerging theme in the environmental science literature is the problem of the science-policy divide. As Folke et al. (2007) note, one of the characteristics of the modern world is the disconnection between production of knowledge and its application. Scientific
evidence rarely becomes a core driver of policy (Campbell et al. 2007, Holmes & Clark 2008). In general, the studies conducted on ‘evidence-based’ policy initiatives reveal that peer-reviewed literature has limited influence on the policy-making processes and management practices (Campbell et al. 2007, Holmes & Clark 2008). The important determinants of knowledge applicability and integration are communication and ongoing interactions between scientists, policy-makers and other stakeholders in both the knowledge-generation and policy-making processes (Roux et al. 2006, Holmes & Clark 2008). In this context, important aspects of institutional analysis can be rules prescribing decision-making processes and defining roles and responsibilities of knowledge holders in the policy-making process. How the ‘science-management’ integration is approached in the literature will be examined in the next part of this chapter.

3.3 Ecosystem management or how to protect

Ecology, conservation biology and related environmental science disciplines define the concept of biodiversity and are the core drivers of the knowledge base. However, a scientific platform alone is not sufficient to address the problem. In the 1990s, ‘ecosystem management’, ‘ecosystem approach’ or ‘ecosystem-based management’\(^7\) has evolved as a core management paradigm, producing shifts in the ways problems have been framed in the resource management field. As Cortner et al. (1998) argue, ecosystem management requires changes in approaches to nature, science and policy and, consequently, changes in existing institutional systems. To gain an understanding of the institutional dimension of ecosystem management, this part briefly reviews its core elements and defined problem attributes. In particular, it examines the concept of adaptive management as a structured management approach widely advocated for the management of complex environmental problems.

3.3.1 Introduction to ecosystem management and problem attributes

Ideas regarding a systems approach to the management of the environment emerged in conservation science in the 1930s and 1940s (Grumbine 1994). As a paradigm shaping environmental and resource management policies, ecosystem management gained acceptance in the late 1980s and early 1990s. In 1995, the Conference of the Parties (COP) to the Biodiversity Convention adopted the ecosystem approach as the primary framework for action towards the goal of sustainable use and conservation of biological diversity (COP 1995 Decision II/8). In 2000 the COP endorsed the description of the ecosystem approach as ‘a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way’, and recommended principles guiding its application (COP 2000 Decision V/6). According to this decision, the ecosystem approach does not preclude the

7 Note: while some sources distinguish between ecosystem management and ecosystem-based management (Slocombe 1998a, Yaffee 1999), these terms will be used interchangeably.
application of other management approaches, such as biosphere reserves, protected areas and others, ‘but could, rather, integrate all these approaches and other methodologies to deal with complex situations’ (COP 2000 Decision V/6). In 2004 the COP introduced guidelines to facilitate implementation of the ecosystem approach (COP 2004 Decision VII/11).

Ecosystem management is variously described and defined. The understanding of the new management paradigm formed a significant part of the academic debate in the 1990s (Grumbine 1994, Christensen et al. 1996, Lackey 1998, Brussard et al. 1998, Cortner & Moote 1998, Slocombe 1998a, 1998b, Yaffee 1999). One of the first comprehensive summaries was published in 1994 by Grumbine, who distilled ten dominant themes characterizing ecosystem management: hierarchical context, ecological boundaries, data collection, monitoring, adaptive management, interagency cooperation, organisational change, humans embedded in nature and values. Grumbine (1994:31) described ecosystem management as a management process which ‘integrates scientific knowledge of ecological relationships within a complex sociopolitical and values framework toward the general goal of protecting native ecosystem integrity over the long term’. The subsequent Report of the Ecological Society of America on the scientific basis for ecosystem management (Christensen et al. 1996:668-9) described ecosystem management more specifically as: ‘management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and processes necessary to sustain ecosystem structure and function.’

Since then, a significant effort has been made to eliminate confusion regarding the practical application of ecosystem management (Yaffee 1999). Several studies have been undertaken to summarise the core elements and principles characterising ecosystem management, both generally and in the context of specific management problems (Grumbine 1994, Christensen et al. 1996, Cortner & Moote 1998, Lackey 1998, Slocombe 1998a, Arkema et al. 2006). Nevertheless, while sharing core themes, the number of elements distinguished and levels of detail differ significantly (Cortner & Moote 1998, Arkema et al. 2006).

To avoid confusion this study predominately builds on the structure of the Report of the Ecological Society of America which distinguishes between eight components of ecosystem management: ‘(1) long term sustainability as the fundamental value, (2) clear operational goals, (3) sound ecological models and understanding, (4) understanding complexity and interconnectedness, (5) recognition of the dynamic character of ecosystems, (6) attention to context and scale, (7) acknowledgement of humans as ecosystem components, and (8) commitment to adaptability and accountability’ (Christensen et al. 1996:669). These elements are briefly outlined below.

Ecosystem management is anthropocentric. Recognising humans as part of ecosystems, linkages between ecosystems and human societies and accommodation of human needs lie at the core of the concept (Grumbine 1994, Christensen et al. 1996, Szaro et al. 1998b). According to
Christensen et al. (1996) intergenerational sustainability is the central goal or value. In the resource management context this goal is widely interpreted as the use which sustains the delivery of environmental goods and services or maintains ecosystem integrity or health (Grumbine 1994, Christensen et al. 1996, Yaffee 1999). At the same time, it is also recognised that management goals and objectives are a matter of societal choice and that values and are ‘determined through negotiations and trade-offs among stakeholders having different perceptions, interests, and intentions’ (SCBD 2004:8).

Ecosystem management evolved in response to the failure of traditional fragmented management to prevent resource depletion and loss of biodiversity (Grumbine 1994). Recognition of the complexity and connectedness of ecological systems is integral to the approach. As Szaro et al. (1998b:2) note, ecosystem management ‘implies that the whole system, or integrated ecological unit, is the context for management, rather than just its individual parts.’ Therefore, in the management field the emphasis is placed on measures facilitating cross-boundary and cross-sectoral interactions of various management bodies, stakeholder involvement, public participation and collaborative decision-making (Grumbine 1994, Szaro et al. 1998b, Cortner & Moote 1998, SCBD 2004).

Ecosystem management is place-based and involves delineation of management boundaries. In the management context, however, ‘ecosystem’ does not imply a single management scale (Christensen et al. 1996). As Yaffee (1999:715) points out, the concept is used ‘more as a mental construct suggesting complexity and systems interactions than a real ecosystem.’ Therefore, ‘ecosystem’ can refer to different functioning units determined by the characteristics of a particular environmental problem and its context (Christensen et al. 1996, Lackey 1998, COP 2000 Decision V/6). As Stanford and Poole (1998:743) suggest, the ecosystem ‘must incorporate, at a minimum, the area necessary to address the largest ecological process of concern.’

‘Understanding’ of complexity, interconnectedness and the dynamic character of ecological systems, and incorporation of this knowledge in decision-making, is the dominant theme in ecosystem management (Grumbine 1994, Christensen et al. 1996, COP 2000). According to the Biodiversity Convention, an ecosystem approach ‘is based on the application of appropriate scientific methodologies focused on levels of biological organization, which encompass the essential structure, processes, functions and interactions among organisms and their environment’ (COP 2000, Decision V/6). Of particular importance is transdisciplinarity and incorporation of findings from various environmental science disciplines, such as ecology, landscape ecology, conservation biology, as well as linkages between environmental and social sciences (Christensen et al. 1996, Cortner & Moote 1998, SCBD 2004). Furthermore, ecosystem management recognises the importance of other sources of environmental information such as indigenous and local knowledge (COP 2000 Decision V/6 Principle 11). As Cortner et al.
(1998:163) summarise: ‘knowledge is recognised as having a social character; science and knowledge are viewed as shaped by society as a whole.’

Building on systems theory, ecosystem management offers new frames of thinking about the functioning of the natural environment and the way it responds to human disturbances. Despite the differences in the operationalisation of the concept, a significant degree of consensus exists regarding the core attributes of ecosystems as management units. According to Christensen et al. (1996:666) ecosystem management is based on ‘four fundamental scientific precepts: (1) critical relevance of spatial and temporal scales, (2) dependence of ecosystem function on its structure, diversity and integrity, (3) ecosystem dynamics in space and time, and (4) uncertainty, surprise and limits to knowledge.’ Each precept has several implications for management.

Ecosystem management characterises ecosystems as complex and open systems which function at a range of spatial and temporal scales, and exchange organisms, matter and energy (Christensen et al. 1996, SCBD 2004). While the emphasis is placed on the delineation of boundaries based on ecological criteria, it is recognised that there is no single appropriate scale for management (Christensen et al. 1996). Furthermore, management may require simultaneous consideration of multiple spatial, temporal and organisational scales (Szaro et al. 1998b, SCBD 2004). As environmental effects may appear at a distance from the place of intervention, or accumulate with other effects, management must consider connectivity or cross-boundary linkages, such as impacts on adjacent or downstream systems or systems linked by migratory species (SCBD 2004). On the temporal scale, management must account for time lags between the management intervention and ecosystem response and incorporate long term objectives (SCBD 2004).

Ecosystem management recognises biological diversity as a property of ecosystems which plays a critical role in producing ecosystem services and provides for both ‘stability (resistance to) and recovery (resilience from) disturbances that disrupt important ecosystem processes’ (Christensen et al. 1996:672). According to the principles of the Biodiversity Convention the priority target is the conservation of ecosystem structure and functioning, as ‘resilience depends on a dynamic relationship within species, among species and between species and their abiotic environment, as well as the physical and chemical interactions within the environment’ (COP 2000 Decision V/6). Therefore, while biodiversity preservation is not a central goal or value, it is pursued to ensure sustained delivery of essential goods and services (Christensen et al. 1996, Lackey 1998).

Ecosystems are dynamic. They are characterised as non-linear self-organising systems which change through time and in response to disturbances both natural and human (Christensen et al. 1996, Dale et al. 2000, COP 2000 Decision V/6). As Holling (1996:733) notes, ecological change does not occur in a linear fashion but is rather ‘episodic, with slow accumulation of natural capital such as biomass and nutrients, punctuated by sudden releases and reorganisation of that capital as the result of internal or external natural processes or of man-imposed
disturbances.” Consequently, there is no single ‘correct’ state of the ecosystem, and management cannot be based on the assumption of some constant desirable state maintained over time (Christensen et al. 1996). Furthermore, natural disturbance regimes are an integral element of the system which needs to be incorporated in management practices (Dale et al. 2000, SCBD 2004). It is recognised that ecosystems have a limited capacity to buffer human disturbances, and planned interventions must, therefore, consider functional limits and apply a precautionary approach (SCBD 2004).

Ecosystem dynamics is linked to two other problem attributes: uncertainty and surprise. The complexity and dynamism of both ecological and social systems often makes uncertainty unavoidable and unlikely events may occur (COP 2000 Decision V/6). From the science perspective, Christensen et al. (1996) distinguish between three categories of uncertainty: true surprises (which cannot be predicted), lack of ecological understanding and guiding principles, and poor data quality and sampling bias. To address the two latter categories, management requires ongoing research, monitoring and modelling, a transdisciplinary approach, as well as communication and information sharing (Christensen et al. 1996, Szaro et al. 1998a). It is also recognised that knowledge is incomplete and decision-making has to be carried out under conditions of uncertainty and a lack of scientific knowledge (SDBD 2004). Adaptive management, which is the core management response proposed to frame decision-making interactions under conditions of uncertainty, will be examined in more detail in the next part of this section.

In summary, ecosystem management focuses on linkages between social and ecological systems with the main purpose of accommodating human needs within the ecological constraints. The paradigm builds on characteristics of ecosystems as complex, dynamic systems interacting at various temporal and spatial scales with uncertainty and surprise as inherent parts of their behaviour. Implementation of ecosystem management requires consideration of multiple values and interests, complex interactions between multiple biotic and abiotic components of the ecological systems, reconciliation of different spatial and temporal scales, recognition of uncertainty and limitations of the current level of knowledge, as well as continuous research and learning.

3.3.2 Adaptive management: decision structure and implementation

According to the Biodiversity Convention (COP 2000 Decision V/6) the ecosystem approach ‘requires adaptive management to deal with the complex and dynamic nature of ecosystems and the absence of complete knowledge or understanding of their functioning’. Adaptive management is the most common proposition made in both academic and political literature on ‘how to’ structure the management of complex environmental problems. Furthermore, it is also increasingly advocated as the core approach to climate change adaptation (Arvai et al. 2006, Lawler 2009). From an institutional perspective, adaptive management can be viewed as a
‘template of rules’, which needs to be integrated into the regulation of management activities. Therefore, this section briefly reviews what kinds of rules or behavioural prescriptions underpin adaptive management, and to what extent they have been, or could be, integrated in institutional designs across problems and contexts.

Introduction to adaptive management

Adaptive management was introduced in the natural resource management field in the mid 1970s by a group of scholars building on ideas gained from control process theory and operations research and management science (Holling 1978, Walters & Hilborn 1978). According to Walters (2007:304) the concept arose from ‘frustration in attempts to use computer modelling to integrate scientific knowledge so as to make useful predictions for decision-makers’. In response to significant gaps in scientific knowledge, ecologists proposed a management concept which treats planned interactions with the natural systems as experiments, and uses intervention outcomes, both anticipated and surprises, as an opportunity for reducing uncertainty (Lee 1993). Information is fed back into the management process to accelerate learning and refine interventions (McLain & Lee 1996). Thus, the process is expected to lead to two beneficial outcomes: improved understanding of the system and improved management (Williams 2011).

While broadly described as ‘learning by doing’, adaptive management is a structured decision-making process. In academic literature the process tends to be pictured as an iterative cycle consisting of several sequential steps. While the description, number and sequence of separate elements can vary (see McFadden et al. 2011 for a review), the core elements of the cycle include: (1) stakeholder engagement; (2) definition of the management problem and management objectives; (3) selection of models representing understanding of the ecological (and social) systems; (4) identification of uncertainty and alternate hypothesis; (5) planning and implementation, (6) monitoring and re-evaluation and (7) adjustment of objectives, models, monitoring or implementation scope or approaches (Williams et al. 2009, McFadden et al. 2011, Rist et al. 2012). In addition, Williams et al. (2009) distinguish between two separate phases of adaptive management: ‘set-up’ and ‘iterative’. The ‘set-up’ phase involves stakeholder interactions with a purpose to define the problem, select objectives, models, management scope and monitoring systems. The ‘iterative’ phase involves decision-making, monitoring and assessment as an ongoing cycle (see Figure 4).
Adaptive management is grounded in two major assumptions: insufficiency of environmental knowledge and the ability of society to learn from interactions with the environmental systems (Lee 1999). Therefore, engagement of stakeholders, scientists and other knowledge holders lies at the core of the process. As McLain and Lee (1996) note, adaptive management relies on the ability of involved interdisciplinary teams of scientists, managers, policy-makers and other stakeholders to jointly identify the management problem, and to select quantifiable management objectives and a set of policy options for implementation. Apart from learning, this process also enhances implementation capacity through resource allocation, and allows recognition of different perspectives and values (Williams et al. 2009). However, as Williams et al. (2009) emphasise, adaptive management is not a conflict resolution mechanism, and its core focus is on learning and shared understanding as the basis for agreement.

Established objectives play a crucial role in performance evaluation and learning (Williams 2011). While ecosystem management incorporates normative concepts such as sustainability, ecosystem health and integrity as overarching goals (Calicott et al. 1998), adaptive management requires their translation into specific and measurable operational objectives (Christensen et al. 1996, Williams 2011). There are no common guidelines about how such objectives should be formulated, but some advice can be found in the literature. For example, Christensen et al. (1996:680) propose stating objectives in terms of ‘desired future behaviour’ of ecosystems or their properties (e.g. status of species, population sizes, productivity) that relate to specific measurements for monitoring. Williams et al. (2009) place emphasis on five general characteristics: specific, measurable, achievable, results-oriented and time-fixed. Furthermore, as the project may involve multiple objectives they also recommend accounting for their relationships ‘so that potential tradeoffs can inform decision making’ (Williams et al. 2009:51).

Figure 4 Two-phase model of adaptive management cycle
Source: reprinted from Williams (2011:1348)
Adaptive management is designed to mimic a scientific approach through identifying uncertainties, specifying and evaluating hypotheses, and selecting actions to test those hypotheses (Gunderson 1999). Alternative hypotheses about the system’s behaviour are translated into models, which represent understanding of the system (Christensen et al. 1996). Models play a crucial role in integrating scientific and other knowledge in decision-making. They are employed to predict impacts of planned management interventions, to select appropriate strategies and identify indicators (Williams et al. 2009). Monitoring systems provide ongoing feedback on the operation of management policies and techniques and indicate what changes are required (Szarö et al. 1998a). According to Williams (2011) effective monitoring provides data for four key purposes: progress evaluation, determination of the state of the resource, improvement in understanding of the resource dynamics, and development or refinement of models. To fulfil this purpose, monitoring needs to reflect key resource parameters and consider appropriate time scales for changes in variables (SCBD 2004, Williams 2011).

The implementation also follows an iterative process where, at each decision point, managers select actions based on established objectives, available alternatives and identified state of the system (Williams et al. 2009). On the operational (i.e. ‘iterative’) level management actions are adjusted based on both monitored response and evolving understanding of the environmental system. The results of management experimentation are fed back into the policy-making or ‘set-up’ process (see Figure 4 above). This, in turn, results in required adjustment or change in objectives, sets of actions, underpinning models or monitoring systems. Adjustments may also be made as a result of changing stakeholder perspectives, or in response to changes in the wider socio-political environment (Williams 2011).

Implementation challenges

While adaptive management has been widely promoted as a ‘panacea’ (Rist et al. 2012) for natural resource management and biodiversity conservation, its application confronts several challenges. Despite the popularity of the concept, the academic literature reports limited success. As several recent reviews suggest (Linkov et al. 2006, McFadden et al. 2011, Rist et al. 2012), there is a limited number of studies reporting implementation of adaptive management, and only a few projects include all core elements. Furthermore, there is a growing recognition of various ecological, institutional and socio-political constraints that impede adaptive management. These are briefly outlined below.

Adaptive management is not a feasible option for all ecosystem management problems characterised by uncertainty and limited knowledge. As Williams et al. (2009) point out, adaptive management requires relatively high control over the relevant variables. Ecosystems are dynamic and may respond to other drivers outside the control of the management intervention, or changes may be too rapid for the designed monitoring systems. As some studies indicate, the attempts to manage large scale systems with complex social-ecological interactions have faced
the problem of limited capacity to consider or control all relevant variables (Norgaard et al. 2009). Incorporation of controllability as another determinant has revealed that in many cases scenario planning can be employed as a better alternative (Wollenberg et al. 2000, Peterson et al. 2003). Scholars also suggest that in cases with low risk and uncertainty resource management can be based on a resource optimisation approach, whereas low uncertainty but high uncontrollability may require hedging strategies (Peterson et al. 2003, Allen & Gunderson 2011).

Apart from ecological factors, the application of adaptive management has been also constrained by the nature of management interventions and their context. As Williams et al. (2009) note, in some cases management intervention leaves ‘no space’ for corrective action. Therefore, while learning still can occur, a lack of opportunity for follow-up adaptation makes adaptive management infeasible. Such constraints can be also placed by existing institutional frameworks precluding ongoing adjustments in the decision-making (see Doremus 2001, Ruhl 2005 for a discussion). As Doremus (2001:55) summarises:

Our dominant paradigm for regulation of private land development is one-time review prior to a proposed action. Proposals for timber harvests, subdivision development, or wetlands filling are either approved or disapproved. If they are approved, we are accustomed to that being the end of story. We have very little history of continuing oversight of private land management, requiring changes over time of our preliminary assessment of the likely environmental impacts proves inaccurate.

Adaptive management has been also confronted by various behavioural and political barriers. On the decision-making level, the implementation has been hindered by the tendency to avoid uncertainty and risk (Allan & Curtis 2005, Allan et al. 2008). For example, Allan and Curtis (2005) in their study exploring the application of adaptive management in regional natural resource management in Australia have revealed a management culture valuing action, control, comfort and clarity. On the organisational level, barriers have been created by established decision-making processes precluding or discouraging flexibility and experimentation, as well as a lack of mechanisms for the implementation (Lee 1993, Allan & Curtis 2005, Jacobson et al. 2006, Williams et al. 2009). Application has been also hindered by conflicting goals (Norgaard et al. 2009), as well as expected cost and benefit calculations (Failing et al. 2004).

On the political level, adaptive management confronts dynamic and unstable political environments (Williams et al. 2009). As Dovers et al. (1996:1159) note, the assumption of continuity and longevity of institutional, policy and management arrangements ‘to match the temporal scales of the natural systems being addressed’ is rarely met. In government agencies experimentation and application of acquired environmental knowledge has been also avoided to escape conflicts or changes in established management programs (Doremus 2001, Ruhl 2005).

**Interpretation**

The critical debate on the conceptualisation and application of adaptive management is only evolving (Allen & Gunderson 2011, Rist et al. 2012). From the institutional perspective, limited
implementation and reported challenges raise two further questions: to what extent adaptive management frames operational decision-making in practice and why it is so popular.

Wide applicability and political appeal so far can be explained by diverse interpretations. A distinction is frequently made between so called ‘active’ and ‘passive’ adaptive management. The term ‘active’ adaptive or ‘scientific’ adaptive management is employed to describe the process which is designed to ‘actively pursue the reduction of uncertainty through management interventions’ (Williams 2011:1350). ‘Passive’ adaptive management has a broad range of interpretations. Some sources describe it as a management process which focuses on the achievement of management objectives with learning as an unintended outcome (Rout et al. 2009, Williams et al. 2009). Others describe it as management which is based on historical best practices and policies followed by review (Allan & Curtis 2005, Lawler 2009).

Furthermore, while in academic literature ‘active’ and ‘passive’ adaptive managements tend to be distinguished from two other types of learning: ‘trial and error’ and learning that does not incorporate systematic monitoring and assessment (Lee 1999, Williams 2011), this distinction is not strictly followed (Rist et al. 2012). In practice, following broad notions of ‘learning by doing’ the term ‘adaptive management’ has been applied to describe any process of learning and resulting decision-making adjustments. For example, Doremus (2001:53) argues that taking into account the need for learning ‘any process that facilitates learning from management decisions and application of that newly-acquired knowledge in the next round of decision-making can properly be referred to as adaptive.’ Furthermore, building on this notion, the concept has been also extended to characterise particular collaborative governance strategies (Dietz et al. 2003, Folke et al. 2005), as well as to describe societal learning (Dovers 1999, 2001b). How to approach institutional dimension of ecosystem and adaptive management is discussed next.

**3.3.3 Ecosystem management: understanding the institutional dimension**

The complexity and scale of ecological problems, diversity of involved actors, as well as different priorities and values suggest that institutions, whether formal or informal, play significant roles in ecosystem management. It is recognised that implementation of ecosystem management requires significant shifts in human behaviour, and consequently changes in existing institutional frameworks (Grumbine 1994, Brussard et al. 1998, Cortner & Moote 1998, Cortner et al. 1998). Some scholars (Meffe et al. 2002) picture ecosystem management at the intersection of three core dimensions: ecological, socioeconomic and institutional. However, while the need for institutional change is frequently stated in the literature this dimension is rarely addressed in a systematic way (for an exception see Cortner et al. 1998).

Initial attempts to systematically review a large body of academic literature addressing institutional aspects of ecosystem and adaptive management encountered several barriers. The first, and perhaps the biggest barrier to a shared understanding, is the application of the term
‘institution’. While the term has been widely employed⁸, it is used extremely broadly and is rarely defined clearly. ‘Institution’ has been employed to describe beliefs, attitudes and values, organisations as decision-makers, informal and formal organisational rules, as well as informal and formal rules regulating broader social interactions (Stankey et al. 2005, Jacobson et al. 2006). Second, significant confusion exists regarding the scope of the term. For example, the review by Jacobson et al. (2006:1519), summarising the barriers to adaptive and ecosystem management identified in the literature, under the category ‘institutional’ lists such aspects as agency culture, organisational environment and inflexible decision-making frameworks. However ‘a lack of clear timelines, goals, and objectives’ is placed in the ‘logistical’ category. Furthermore, while the emphasis tends to be placed on such ‘management-related’ institutional barriers as problems with funding allocation (Jacobson et al. 2006, Allan et al. 2008), ‘ecosystem management-related’ aspects, such as established operational objectives, prescribed management actions and environmental indicators, are rarely recognised as rules or institutions.

Despite the above problems, the literature does contain a variety of propositions and findings which have important implications for institutional design and analysis. Several studies have made an attempt to distil a range of general elements or principles characterising ecosystem management as the core criteria for the analysis of selected institutions (Brody 2003, Arkema et al. 2006, Shandas et al. 2008). This approach, however, encounters one major pitfall. It implies that ecosystem management is based on some common or generic set of principles that need to be considered in institutional designs. Struggles with the implementation of adaptive management, which tends to be incorporated as one of the core principles or themes (Grumbine 1994, Christensen et al. 1996, Shandas et al. 2008), however indicate that such propositions need to be approached with caution. This is also reflected in the Biodiversity Convention guidelines, issued to support the principles of the ecosystem approach, which suggest diversity of solutions that ‘may’ need to be considered in institutional design (SCBD 2004).

To deal with the institutional dimension of ecosystem management this study proposes another solution. It builds on two approaches, which can be labelled as ‘problem attribute-based’ and ‘structure-based’. The first approach revolves around the identification of specific problem properties/attributes which have implications for institutional designs.⁹ The second distinguishes between the levels in the governance system at which particular aspects of the problem should be regulated.

Changes in the description of the problem attributes introduced by the ecosystem management paradigm (see section 3.3.1) spawned a large diversity of propositions about what changes are required in existing management structures and practices. Drawing on Christensen et al. (1996),

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⁸ Note: search for publications since 1994 using the Web of Sciences search engine under the key-words ‘institutions’ and ‘ecosystem management’ or ‘adaptive management’ in all fields offered more than 29000 peer-reviewed articles.
⁹ Note: this approach also underpins the concept of ‘institution-ecosystem fit’ (Young 2002, Folke et al. 2007), which will be explored in detail in the next section.
‘new’ problem attributes driving this debate could be grouped under four major themes: complexity, connectivity in time and space, uncertainty and dynamics, and humans as ecosystem components. Based on these themes Table 1 (at the end of this section), summarises the review of twelve sources describing elements of ecosystem management. The review also includes several summaries published previously (Grumbine 1994, Christensen et al. 1996, Cortner & Moote 1998). These themes offer a good foundation for structuring both general and issue-specific propositions found in the literature. Linking them to the typology of institutional functions (see section 2.2.3) allows the identification of five broad dimensions which can create opportunities or constraints for ecosystem management. These are briefly outlined below.

The first institutional dimension contains the rules providing for operational guidance (i.e. coordinating and cognitive functions). Ecosystems are complex systems consisting of many interacting properties. This complexity implies both revision and institutionalisation of diverse theories, principles and concepts developed by different disciplines of the environmental sciences. From this perspective, ecosystem management requires continuous design and revision of operational rules prescribing what actions, how, when and under what conditions should be taken, or what environmental factors should be considered to achieve anticipated objectives (see section 3.2.3 for more detailed discussion).

The second dimension contains the rules allocating decision-making authority (i.e. enabling-constraining function). Complexity and connectivity of ecological systems and their properties may require changes in the allocated management scope and scale. These, as Cortner et al. (1998) point out, tend to be divided by political boundaries and resource sectors. Systems dynamics and uncertainty may also require reforming established institutional goals, decision-making timeframes, as well as levels of discretion allocated to management bodies (i.e. ‘flexible’ design). New rules may be required to authorise participation of knowledge holders and the wider public in the decision- and policy-making processes.

The third dimension contains the rules shaping the financial capacity of management bodies or regulatory authorities (i.e. allocating function). Implementation of monitoring programs, design of models, collection of data, and maintenance of data bases requires ongoing allocation of financial resources. Viewed from this perspective, ecosystem management may require changes in rules providing for resource distribution and allocation among the management bodies and across problem areas (Szaro et al. 1998a). As outlined previously, insufficiency of financial resources is one the most commonly reported and recognised institutional barriers for the implementation of ecosystem and adaptive management.

The fourth dimension contains the rules providing for incentives to support conservation and sustainable use of environmental resources (i.e. payoff and normative functions). Whether involved institutional actors will be interested to pursue sustainability or observe functional limits largely depends on rules shaping incentives. From this perspective, introduction of new
values, such as ecosystem services, may require the design of new mechanisms and changes in current market exchange systems. In addition, the rules may need to be altered to create incentives for the management bodies to address cross-boundary effects, cooperate or share information and knowledge (Cortner et al. 1998, Imperial 1999).

Finally, the fifth dimension consists of the diverse configurations of rules framing decision-making processes (i.e. structuring function). Implementation of ecosystem management may require institutional change to enable interactions between organisations involved in the management of separate components of shared ecological systems (e.g. cross-sectoral coordination). As Imperial (1999:452) points out, ecosystem management ‘can be seen as explicit attempt to build, manage, and maintain interorganisational networks, in other words, to develop an institutional ecosystem.’ Changes may also be required in rules for decision- or policy-making arenas to enable incorporation of different considerations, deliberation, collective learning or conflict resolution.

Institutional systems are hierarchical. In the adaptive management context Williams et al. (2009) distinguish between two levels of management interactions (see section 3.2.2). From an institutional perspective this model also implies a hierarchy of two groups of interacting rules shaping ‘on ground’ outcomes. According to the distinction, the ‘set-up’ level rules provide the framework structuring interactions between responsible agencies, stakeholders and other actors in the rule-setting or policy-making process. The ‘implementation’ level or operational rules directly guide management decisions affecting ‘on ground’ outcomes. While this model has been applied to specifically describe adaptive management, it can be argued that it largely reflects the hierarchy of rules guiding most ecosystem management situations (see Figure 5).

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**Figure 5** Two levels of rules structuring implementation of ecosystem management

Source: adapted from Williams et al. (2009:38), institutional factors added by author
Adding the rules hierarchy to the functional domains identified above helps to locate particular groups of rules within the system and to examine their influence. For example, involvement of new institutional actors such as scientists, local or indigenous communities and other stakeholders, widely promoted in the literature (see Table 1), occurs at the rule setting stage (i.e. ‘set-up’ level) rather than at the implementation level. In some contexts, the rules specifying participation at this level may affect geographical, temporal and functional scales addressed in the management arrangement (see Williams et al. 2009 for a discussion). ‘Implementation’ level rules may specify objectives, the allowable scope of actions, enabling or constraining ecological conditions, or provide other guidance. To what extent these rules make ‘ecological sense’ in relation to the systems, or their properties under consideration, directly affects management results.

Both ‘problem-’ and ‘structure-’ based approaches can assist in structuring the analysis of a large diversity of institutional designs providing for human-environment interactions. More detailed insight into the several dimensions structuring relations between different actors of the environmental governance are provided in the next section.
<table>
<thead>
<tr>
<th>Problem elements</th>
<th>General recommendations</th>
<th>Specific recommendations for management</th>
</tr>
</thead>
<tbody>
<tr>
<td>Complexity</td>
<td>Consideration of multiple levels of the biodiversity hierarchy (^{2,10})</td>
<td>Hierarchical set of goals and objectives, supported by targets, indicators and monitoring (^{6})</td>
</tr>
<tr>
<td></td>
<td>Consideration of linkages between separate components of the system (^{1,2,3,4,5,7,9})</td>
<td>Management based on available scientific knowledge (^{1,2,3,4,5,7,8,9})</td>
</tr>
<tr>
<td></td>
<td>Conservation of ecosystem structure and function, to maintain ecosystem services (^{2,10})</td>
<td>Data base on ecosystem structure, composition, and processes at appropriate spatial scales (^{4,5})</td>
</tr>
<tr>
<td></td>
<td>Hierarchical set of goals and objectives, supported by targets, indicators and monitoring (^{6})</td>
<td>Continuous improvement of understanding of the response of ecosystems, research and data collection to support management (^{2,10})</td>
</tr>
<tr>
<td></td>
<td>Management based on available scientific knowledge (^{1,2,3,4,5,7,8,9})</td>
<td>Application of ecological concepts and principles in the decision-making (^{3,7})</td>
</tr>
<tr>
<td></td>
<td>Data base on ecosystem structure, composition, and processes at appropriate spatial scales (^{4,5})</td>
<td>Cross-sectoral cooperation and coordination of management activities (^{10})</td>
</tr>
<tr>
<td></td>
<td>Continuous improvement of understanding of the response of ecosystems, research and data collection to support management (^{2,10})</td>
<td>Application of various information sources, including indigenous and local knowledge (^{1,10})</td>
</tr>
<tr>
<td></td>
<td>Application of various information sources, including indigenous and local knowledge (^{1,10})</td>
<td>Monitoring of population sizes of vulnerable and economically important species (^{10})</td>
</tr>
<tr>
<td>Connectivity in space and time</td>
<td>Scale-dependent management, boundaries defined based on the characteristics of particular environmental problem (^{1,2,3,4,7,8,9,10})</td>
<td>Management boundaries clearly defined (^{3,5,6})</td>
</tr>
<tr>
<td></td>
<td>Consideration of multiple management scales (^{1,3,5,7})</td>
<td>Boundaries for management defined operationally by users, managers, scientists and indigenous and local peoples (^{10})</td>
</tr>
<tr>
<td></td>
<td>Consideration of cross-scale linkages or management effects on adjacent and other ecosystems (^{3,5,10})</td>
<td>Place- or region-based objectives, with scopes and approaches defined appropriately for each given situation (^{7})</td>
</tr>
<tr>
<td></td>
<td>Recognition of varying temporal scales, including lag-effects that characterize ecosystem processes (^{3,10})</td>
<td>Long term management goals and objectives and long term commitment (^{3,10})</td>
</tr>
<tr>
<td></td>
<td>Recognition of varying temporal scales, including lag-effects that characterize ecosystem processes (^{3,10})</td>
<td>Monitoring systems designed to accommodate the time scale for change in the ecosystem variables selected for monitoring (^{10})</td>
</tr>
<tr>
<td></td>
<td>Scale-dependent management, boundaries defined based on the characteristics of particular environmental problem (^{1,2,3,4,7,8,9,10})</td>
<td>Readjustment of scale of institutional response to coincide more closely with spatial and temporal scales of processes in the area under management (^{10})</td>
</tr>
<tr>
<td></td>
<td>Consideration of cross-scale linkages or management effects on adjacent and other ecosystems (^{3,5,10})</td>
<td>Landscape as a scale for the management of human-environment interactions (^{3,4,7})</td>
</tr>
<tr>
<td></td>
<td>Recognition of varying temporal scales, including lag-effects that characterize ecosystem processes (^{3,10})</td>
<td>Cooperation and coordination of management actions across administrative and political boundaries (^{3,10})</td>
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<td></td>
<td>Management boundaries clearly defined (^{3,5,6})</td>
<td>Consideration of cross-boundary implications in impact assessments (^{10})</td>
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<td></td>
<td>Boundaries for management defined operationally by users, managers, scientists and indigenous and local peoples (^{10})</td>
<td>Involvement of stakeholders and technical experts to consider how to minimize adverse consequences on connected elements (^{10})</td>
</tr>
<tr>
<td></td>
<td>Place- or region-based objectives, with scopes and approaches defined appropriately for each given situation (^{7})</td>
<td>Consideration of the potential offsite impacts in environmental and strategic impact assessments (^{10})</td>
</tr>
<tr>
<td></td>
<td>Long term management goals and objectives and long term commitment (^{3,10})</td>
<td>Introduction of national and regional feed-back mechanisms to monitor the effects across ecosystems (^{10})</td>
</tr>
<tr>
<td></td>
<td>Monitoring systems designed to accommodate the time scale for change in the ecosystem variables selected for monitoring (^{10})</td>
<td>Capacity enhancement to analyse and understand the temporal and spatial scales (^{10})</td>
</tr>
</tbody>
</table>
## Dynamics and Uncertainty

Ecosystems are dynamic systems and the management must recognise that the change is inevitable\(^1,3,8,10\). Traditional disturbance regimes may be important for ecosystem structure and functioning\(^10\). Uncertainty and surprises cannot be eliminated\(^3,8,10\). Knowledge base is provisional, incomplete, and subject to change\(^3\).

Focus on the maintenance of natural ecological processes\(^3,10\). Application of adaptive management practices to improve and correct management decisions\(^1,3,4,5,7,8,9,10\). Research and knowledge update to reduce uncertainty and fill knowledge gaps\(^3,4\). Management aimed at maintenance or restoration of ecosystems’ resilience\(^10\). Application of precautionary approach\(^1,10\).

Measurable objectives and targets\(^1,3\). Monitoring of changes in various components of ecosystems linked to management goals and objectives\(^1,2,4,5,7,10\). Monitoring programs and reliable set of indicators for ecosystem attributes\(^1,3,5\).

Scenario analysis\(^1\). Application of ecosystem models and modelling tools and programs (e.g., GIS) as a decision-making support\(^1,3,5\). ‘Uncertainty’ or ‘risk’ analysis to select management actions\(^4\). Management capacity to adapt to new information and knowledge\(^2,3,8,10\). Ongoing interaction between scientists, managers and public\(^2,10\). Utilisation of traditional knowledge and practices\(^10\).

Information distribution, awareness building and communication\(^1,3,10\).

### Humans as Ecosystem Components

Ecosystems are made up of socioeconomic and biophysical components and their interactions\(^2,3,7,8,10\). Humans are fundamental influences on ecological patterns and processes and are in turn affected by them\(^2\).

Sustainable use as the primary goal and objective\(^1,2,3,7,8,9,10\). Ecosystems must be managed within the limits of their functioning\(^1,3,6,8,10\). The goals and objectives are a matter of societal choice and reflect social values and priorities\(^1,2,3,6,7,8,10\). Desired environmental benefits determined by public and facilitated by scientists\(^4,6,7,8\).

Sustainability, ecosystem integrity, health or viability as overarching strategic management goals\(^1,2,5,10\). Development of strategic management goals which are socially and politically possible and consistent with the existing legal framework\(^3\). Involvement of stakeholders and public in the goal setting and planning process\(^1,3,5,10\). Intersectoral and interagency cooperation and coordination in the goal setting and decision-making\(^2,3,5,8,10\).

Participatory, consultative, collaborative decision-making incorporating a range of values\(^1,5,7,8,9,10\). Consensus in goal setting and planning\(^1,3\). Organisational and institutional change to enable collective decision-making\(^2,3,7,8\).

Decentralised decision-making arrangements\(^8,10\). Assessment of cumulative effects of interventions over time and space to consider ecosystem limits\(^10\). Interdisciplinary approach to research and management to integrate social and biophysical aspects of the management\(^1,2,3,4,5,7,8,10\). Sharing of technical and scientific information among stakeholders, information transparency and availability\(^10\). Alignment of cost-benefit structure to ensure equitable sharing of costs and benefits from the exploitation of the resources\(^10\). Application of economic valuation methodologies for ecosystem goods and services (direct, indirect and intrinsic values); and for the environmental impacts (effects or externalities)\(^10\). Development of integrated natural resource management systems and practices to ensure the appropriate balance between the conservation and use of biological diversity\(^10\).

**Source:** compiled by author based on Slocombe (1998a,b)\(^1\), Grumbine (1994)\(^2\), Christensen et al. (1996)\(^3\), Stanford and Poole (1996)\(^4\), Brussard et al. (1998)\(^5\), Lackey (1998)\(^6\), Szaro et al. (1998a)\(^7\), Cortner and Moote (1998)\(^8\), Yaffee (1999)\(^9\) COP5 (2000 Decision V/6) \& SCBD (2004)\(^10\).
3.4 Environmental governance and biodiversity or who should protect

Incorporating environmental knowledge into ‘a complex sociopolitical and values framework’ (Grumbine 1994:31) makes processes and structures of the social dimension, such as power sharing, distribution of roles and responsibilities, an integral part of the biodiversity protection problem. These interactions are often explored under an overarching concept of ‘governance’ or, more specifically, ‘environmental governance’. The concept of ‘environmental governance’ is broad and covers multiple levels of social organisation (e.g. international, national, local). While interconnected, each level consists of different sets of actors and configurations of power relations. Therefore, this section examines the structure and processes of ‘local environmental governance’ that most directly determine ‘on ground’ environmental outcomes.

3.4.1 Local environmental governance: centralised, decentralised or cross-scale

The concept of ‘governance’ has been employed to describe the shift from traditional top-down government by the state to a broader set of rules and management practices emerging from various interactions between state, public and private actors (Rhodes 1996, Buizer et al. 2012). ‘Environmental governance’ is variously defined. For example, Lemos and Agrawal (2006:298) describe it as ‘a set of regulatory processes, mechanisms and organisations through which political actors influence environmental actions and outcomes’. Lebel et al. (2006:2) define it as ‘the structures and processes by which societies share power’ and include ‘laws, regulations, discursive debates, negotiation, mediation, conflict resolution, elections, public consultations, protests, and other decision-making processes.’ The World Resource Institute defines the concept simply as ‘the exercise of authority over natural resources and the environment’ (WRI 2002:2).

As many environmental problems, including biodiversity loss, are place-based, the design and operation of local environmental governance is considered as both the cause of, and the solution to, the expanding set of environmental problems. During the last two decades a significant body of literature has been generated on the design and operation of local environmental governance, and this topic continues to remain at the centre of the political and academic debate. In general, there is an evolving consensus towards a range of ‘good governance’ principles or attributes, such as openness or transparency, participation, accountability, efficiency, effectiveness and equity (UNDP 2002, SCBD 2004, Batterbury & Fernando 2006). At the same time, questions such as - which actors, in what capacity, at what level of social organisation should manage what interactions and with what types of resources, and how their interactions should be structured - are far from answered. Numerous concepts are employed in the academic literature to describe diverse trends in the design and operation of the environmental governance systems, for example: localisation, regionalisation, ‘glocalisation’, socialisation, marketisation, integration,
 decentralised governance, network governance, adaptive governance, multi-level governance, polycentric governance, place-based governance.

This section cannot address the whole complexity of issues raised with regard to the design and operation of local environmental governance. Therefore, maintaining the focus on the question by whom biodiversity should be managed or protected, this section examines various reforms shaping governance structure and the related academic debate. It is based on the two types or modes of governance structures distinguished by Hooghe and Marks (2003) which, drawing on Lemos and Agrawal (2006), can be labelled as ‘decentralised’ and ‘cross-scale’.

**Decentralised governance**

Decentralisation is often emphasised as the key answer to the problems of a top-down approach to the management of human-environment interactions (Peuhkuri & Jokinen 1999). As Lemos and Agrawal (2006) note, decentralisation in the governance of renewable resources has become a characteristic feature of the late 20th and early 21st century. Decentralisation ‘to the lowest appropriate level’ is also supported by the Biodiversity Convention, and is one of the main principles of the ecosystem approach (COP 2000 Decision V/6, Principle 2). The rationale is based on two core assumptions: that ‘decentralised systems may lead to greater efficiency, effectiveness and equity’, and that ‘the closer management is to the ecosystem, the greater the responsibility, ownership, accountability, participation and use of local knowledge’ (COP 2000 Decision V/6). Furthermore, decentralisation tends to be linked to the principle of subsidiarity (Ribot 2002, MEA 2005b) which, as Føllesdal (1998:190) describes, ‘regulates authority within a political order, directing that powers or tasks should rest with the lower-level sub-units of that order unless allocating them to a higher-level central unit would ensure higher comparative efficiency or effectiveness in achieving them.’

Linked to the overall democratisation of policy-making processes, the foundation of decentralised environmental governance lies in the distribution of decision-making authority ‘closer to the people’. However, there is no general agreement on how to determine the ‘lowest appropriate level’ or how to interpret the subsidiarity principle in the environmental context (Ribot 2002, Larson & Ribot 2004, Marschall 2007). The Biodiversity Convention guidelines to the ecosystem approach list several factors that should be considered in choosing an appropriate management body: representation of the appropriate community of interests, management capacity and commitment, efficiency, conflict of interests, as well as the effect on marginalised groups of society (SCBD 2004, guideline 2.5). Furthermore, Principle 7 of the ecosystem approach adds the ‘ecological’ dimension, or requirement to match the ‘temporal and spatial scales’ appropriate to the problem being managed (COP 2000 Decision V/6).

‘Decentralisation’ refers to the transfer of powers from the state to lower level decision-making authorities (Ribot 2002). It incorporates several strategies which can be pursued in authority distribution. So called ‘political’ or ‘democratic’ decentralisation involves a transfer of resources
and power to a lower level of representative authority in an autonomous and discretionary decision-making sphere (Hutchcroft 2001, Ribot 2002). According to Ribot (2002:4), democratic decentralisation ‘aims to increase popular participation in local decision making.’ ‘Administrative decentralisation’ involves two strategies: transfer of powers and responsibilities to the regional or local offices of the state government (deconcentration) (Hutchcroft 2001, Ribot 2002), and transfer of administrative responsibilities to the lower levels of a representative authority (devolution) (Hutchcroft 2001). The term ‘devolution’ has also been applied to describe the transfer of administrative functions to non-state actors such as non-government organisations (Larson & Soto 2008). Finally, another type of decentralisation is the so called ‘marketisation’ or ‘privatisation’, which involves the transfer of management authority to private or market actors (Van Tatenhove & Leroy 2003). These strategies and underpinning assumptions are briefly outlined below.

Democratically elected local governments have been widely promoted as the main level of social organisation for sustainable management of environmental resources. In the international arena, local authorities gained particular attention after the United Nations Agenda 21 (Chapter 28) which emphasised the local level as the main management level responsible for operational outcomes of sustainable development (UNCED 1992). Allocation of powers and administrative responsibilities to elected local authorities has been linked to promotion of democracy, increasing local participation, equity, as well as better mediation of multiple interests (Ribot 2002, Larson & Ribot 2004). Local governments are assumed to better understand and respond to the needs of their constituents (WB 1999). Local authorities have been also promoted as the management level which can better utilise environmental knowledge and arrive at solutions more suitable to particular social and environmental conditions (Andersson & Ostrom 2008).

Another widely supported strategy, particularly in developing countries, is the devolution of authority to new organisational entities. Since the 1990s, increasing interest has been focussed on community- or user-based resource management as an alternative to centralised forms of state control (Armitage 2005). Extensive research worldwide has been conducted on the operation of small-scale common-pool resource (CPR) management systems, including the ability of communities to manage shared environmental resources (Dietz et al. 2003). These have proven the effectiveness of local management systems to share and incorporate ecological knowledge and enforce rules to sustain the resources over long periods of time (Ostrom 1990, Berkes & Folke 1998, Dietz et al. 2003). Community-based decentralisation strategies are built on assumptions of close connection to natural resources, possession of required ecological knowledge, motivation to sustain the resource, as well as the capacity to monitor and enforce compliance (Armitage 2005).

Finally, a parallel trend shaping governance reforms, particularly in developed countries such as Australia, Canada, U.S. and U.K., is the so called ‘marketisation’ or ‘privatisation’ of environmental governance (Meadowcroft 1998, Kettl 2000, Reddel 2002, Evans et al. 2005).
Since the 1980s, many state governments have experienced a decline in the capacity to provide for the increasing scope of public services. Following neoliberal policy direction, they have distributed part of their functions to market and non-profit organisations via contracts and various forms of public-private partnerships (Savas 2000). Many management functions, including environmental management, have been outsourced to corporatized organisations. In some countries this strategy has also involved privatisation of public environmental resources such as water and forests (Bakker 2003, Wilder & Lankao 2006). Marketisation has been linked to such benefits as improved quality and efficiency in the delivery of services, as well as being applied to reduce fiscal pressures (Kettl 2000). As Kettl (2000:493) notes it allows the government to tap into capacities of private actors and to ‘increase its reach without increasing its size.’

In line with neoliberal policies, an increasing trend is the introduction of new mechanisms which adjust individual incentives in favour of environmental outcomes (Lemos & Agrawal 2006). Apart from taxation, proposed incentive-based solutions include resource markets, tradeable emission allowances, tradeable rights, subsidies, as well as voluntary communicative measures such as certification and eco-labelling (Lemos & Agrawal 2006, Schröter-Schlaack & Ring 2011). As Cashore (2002:504) argues, in many cases market based instruments have created ‘non-state-market-driven governance’ where ‘the relatively narrow institution of the market and its supply chain provides the institutional setting within which governing authority is granted and through which broadly based political struggles occur.’ Under market conditions compliance is expected to result from market incentives (Cashore 2002).

Much of the democratic decentralisation literature focuses on two aspects of effective environmental governance. The first is the (re)distribution of the decision-making power and administrative responsibilities. In this context, the common area of concern is power relations between the state and lower level governance units in the political-administrative hierarchy. Among common identified barriers to effective decentralised governance are resistance of the state governments, insufficient decentralisation of decision-making power, and devolution of management responsibilities without allocation of required resources or capacity building (MEA 2005b, Ribot et al. 2006, Larson & Soto 2008). The second is social outcomes resulting from interactions between local decision-making bodies and their constituents. In this regard, particular emphasis is placed on two attributes of governance: public participation and accountability (Ribot 2002, Larson & Ribot 2004). Of major concern, particularly in developing countries, is the design of accountability mechanisms, as many decentralisation reforms have failed, leading to persistence of centralised or elite control (Batterbury & Fernando 2006).

Accountability and public participation in the rule setting process also dominate the debate between proponents of democratic decentralisation and those supporting marketisation strategies. Distribution of management authority to market actors has been linked to problems of
limited accountability and public control. This, in turn, has been linked to risks of unequal power relations with market actors dictating the rules (Ribot 2002, Batterbury & Fernando 2006).

**Cross-scale governance**

Several arguments have been expressed against full decentralisation as ‘the solution’ to environmental problems. The common argument is that ecological systems, and consequently management effects, cannot be aligned with local administrative or community boundaries. As Folke et al. (2007) note, many local actions may have regional or even global effects. Another factor is the complexity of incentive systems shaping the motivation of local management bodies. Numerous studies have demonstrated that increasing ‘institutional connectivity’ increases vulnerability of decentralised local management systems to perverse market incentives (Dietz et al. 2003, Armitage 2005, Brondizio et al. 2009). As the MEA (2005b:152) acknowledges: ‘the fact that biodiversity generates benefits beyond local and even regional boundaries implies that decentralization of biodiversity conservation management could shift management toward local benefit provision if proper national or international incentives and management structures are not nested with local management.’ Finally, it is argued that the complexity and scope of environmental problems that need to be addressed within a particular geographical area often exceed the capacity of a single management body (Mitchell 2005).

A multi-level and polycentric (i.e. cross-scale) approach has been promoted as another alternative to the design of local environmental governance (Folke et al. 2005, Lebel et al. 2006, Andersson & Ostrom 2008, Armitage et al. 2009, Brondizio et al. 2009). ‘Multi-level’ governance implies that human-environment interactions need to be addressed at multiple levels of social organisation (Brondizio et al. 2009). As Adger et al. (2005:12) summarise:

many, if not all [resource management] systems, are inherently cross-scale and their success in promoting sustained engagement and resilient and shared management are determined by factors at a range of levels from constitutional and organizational to those at the level of resource users.

‘Polycentricity’ characterises the organisational structure of multiple, relatively independent management bodies with authority to make and enforce rules in specified matters and localities (Andersson & Ostrom 2008). Polycentric governance has a flexible organisational structure which is designed ‘with respect to particular policy problems – not particular communities or constituencies’ (Hooghe & Marks 2003:240). Governance structure consists of both general purpose government and management units created to fulfil distinct functions (Andersson & Ostrom 2008). Special purpose sub-units are ‘nested’ within the overall governance system complementing, overlapping or even competing with general purpose jurisdictions (Andersson & Ostrom 2008). They may also include ad hoc, temporary management bodies created to resolve a particular problem, such as commissions, taskforces or agencies (Hooghe & Marks 2003). Consequently, it has been argued that polycentric governance offers flexibility in shifting jurisdictional boundaries, allows for functional specialisation and better accumulation of expertise, as well as providing the advantage of economies of scale (Imperial 1999).
A multi-level and polycentric mode implies that environmental problems are concurrently addressed by several decision-making bodies. In contrast to the decentralised approach, which focuses on optimal power distribution and treats local authorities as independent units, the polycentric perspective focuses on linkages (Andersson & Ostrom 2008). The effectiveness of environmental governance is linked to the collective capacity of involved decision-makers to provide for the management of shared environmental problems. Consequently, one of the effectiveness determinants are strategies which counteract fragmentation and create required linkages between diverse management authorities.

There is a range of ‘integration’ mechanisms promoted in the literature. Among them are informal and formal agreements, boundary or bridging organisations, informal policy networks and co-management arrangements (Cash et al. 2006, Folke et al. 2007). During the last decade increasing attention has been paid to co-operative or collaborative co-management models which link state, community and private sector actors. They range from informal power-sharing arrangements, building on self-organising capacities, to more formal structures supported by legislative frameworks and centralised funding (Folke et al. 2005). Collaborative decision-making built on trust has been linked to improvements in knowledge sharing and social learning, conflict reduction among stakeholders, as well as improved policy outcomes (Conley & Moote 2003, Olsson et al. 2004, Imperial 2005).

The design and operation of co-management arrangements is linked to another concept, that of ‘adaptive governance’ which is gaining increasing recognition in the environmental governance literature (Folke et al. 2005). Adaptive governance has evolved as an umbrella concept to describe diverse formal and informal arrangements designed to resolve a range of cross-jurisdictional problems such as water pollution, resource management (e.g. fisheries, forest, irrigation) and biodiversity protection (Olsson et al. 2004, Folke et al. 2005, Lebel et al. 2006). Adaptive governance relies on collaboration of a diverse set of stakeholders, collaborative generation of environmental knowledge, local knowledge systems, monitoring, and learning from experience and flexible adjustment to changes (Olsson et al. 2004, Armitage 2005).

**Centralised, decentralised or cross-scale?**

Distribution of power and responsibilities for the management of environmental resources and biodiversity between state and local governments, state-private-public interactions and related opportunities and challenges are the subject of extensive political and academic debate, which cannot be fully summarised in this section. Nonetheless, several themes having implications for institutional analysis can be distilled from the literature.

The understanding of different modes of environmental governance is still evolving. At the same time, there is an increasing recognition that each strategy has its strengths and weaknesses, and as such does not provide an ultimate resolution to all environmental problems (Imperial 1999, Larson & Ribot 2004, Andersson & Ostrom 2008). For example, there is a growing body of
empirical studies demonstrating that community-based approaches can fail due to such factors as lack of social capital for self-organisation, lack of interest to protect the resource, conflicting interests and power relationships, as well as perverse external incentives (Dietz et al. 2003, Armitage 2005, Brondizio et al. 2009). Also, democratically elected local governments may face very mixed incentives to perform environmental duties (Flynn 2000, Larson & Soto 2008). Similarly, analysts exploring collaborative co-management arrangements warn against uncritical recommendation of this strategy as a universal cure (Conley & Moote 2003, Imperial 2005, Armitage et al. 2009). The studies emphasise that the diversity of actors, environmental problems and their contexts require application of a combinations of measures tailored to suit particular decision-making situations (Ostrom 2005, 2009, Batterbury & Fernando 2006, Underdal 2010, Young 2011). At the same time, they acknowledge that there is still an insufficient understanding of what modes and strategies are best suited for what problem settings (Armitage et al. 2009, Ostrom 2009, Young 2011).

Problems are encountered in the development of theories to provide sufficient guidance for the design of environmental governance (Ostrom 2010). Studies exploring the implementation of decentralisation strategies in developing countries repeatedly report the failure of blueprint models imposed by international agencies based on normative ideals of good governance (Larson & Ribot 2004, Batterbury & Fernando 2006, Larson & Soto 2008). As Larson and Ribot (2004:7) acknowledge:

Decentralisations are not working as some theories suggest. […] this failure is partly because of the fact that decentralisations are not being implemented, but also due to the factors that democratic decentralisation theories cannot or do not account for.

A growing recognition of the theory-practice gap has also created some tension in the scholarly debate. Some argue for moving beyond the search for ‘panaceas’ and simplified and generalised assumptions, which often underpin promotion of particular modes, strategies or actors as being the most effective (Ostrom 2005, 2009, Armitage et al. 2009, Young 2011). In this context, one of the leading analysts Elinor Ostrom (2005:256) points out that the presumption that complex policy problems ‘can be solved through the adoption of simple designs that are given general names, such as private property, government ownership, or community organisation, is a dangerous academic approach’.

Complexity of environmental governance is a frequent theme. As several authors argue, new management or governance sub-units, such as co-management or public-private partnership arrangements, have not replaced the old government hierarchy but rather are layered on top of it (Meadowcroft 2002, Trubek & Trubek 2007, Gunningham 2009). Furthermore, while state governments are losing a dominating role in the rule setting process, this role has not been shifted to a single decision-making authority, nor fully substituted by market mechanisms (Meadowcroft 2002). As Lemos and Agrawal (2006) conclude, emerging forms of environmental governance are hybrid, multilevel and cross-sectoral in nature. Consequently,
Local environmental governance can be framed by a variety of authority distribution strategies which, in various configurations, can operate simultaneously in any geographical location (see Figure 6). How different modes and strategies interact or fit together, and their environmental implications, still remains to be explored (Lemos & Agrawal 2006).

Figure 6 Actors, strategies and instruments of environmental governance
Source: adapted from Lemos and Agrawal (2006:310), governance actors and instruments added by author

In this regard, one of the emerging concepts offering a new theoretical frame for the analysis of complex human-environment interactions is the concept of ‘social-ecological systems’ (SES). The concept has evolved on the basis of complex systems theory, and is increasingly applied in the analysis of common-pool resource management problems encountered at different levels of social organisation (Berkes & Folke 1998, Anderies et al. 2004, Adger et al. 2005, Folke 2006). According to Anderies et al. (2004:18) a social-ecological system can be defined as ‘an ecological system intricately linked with and affected by one or more social systems’. The concept places emphasis on two properties: interdependencies between people and ecosystems, and complexity of ecological, social and social-ecological interactions.

As complexity makes any SES unique, the analysis of these systems requires the identification of multiple variables of both ecological and socioeconomic systems that define patterns of interactions and the overall outcomes (Carpenter & Brock 2004, Anderies et al. 2006). It is also acknowledged that complex, non-linear SES systems can be unpredictable. This makes the effects of policy interventions uncertain (Carpenter & Brock 2004). How governance analysts approach the institutional dimension is discussed next.
3.4.2 Environmental governance and institutions

In the governance literature, institutions are approached from two perspectives. First, they are discussed as an external framework which enables implementation of particular governance strategies. In this context, the emphasis is placed on the design features performing such functions as: defining the roles of the decision-making bodies, delineating authority boundaries, providing for accountability mechanisms, resource distribution or conflict resolution (Olsson et al. 2004, Armitage 2005). Second, institutions as an internal system are viewed as one of the core determinants of the capacity of established governance units (e.g. community-based management bodies, collaborative management arrangements) to manage environmental resources. In this context, established ‘institutional infrastructure’ is seen as an integral part of a particular system, which determines how involved actors handle or share environmental resources, exchange environmental knowledge, cooperate and make decisions (Ribot 2002, Lebel et al. 2006, Armitage et al. 2009).

Reforms in governance structures have raised interest in institutional design principles which could guide the design and operation of new governance units. According to Ostrom (1990:90), design principles are ‘an essential element or condition that helps to account for the success of […] institutions’. In this context, the most widely known set are the eight design principles enabling the operation of small-scale community-based management arrangements proposed by Ostrom (1990): clearly defined boundaries, proportional equivalence between benefits and costs, collective choice arrangements, existence of monitoring systems, graduated sanctions, conflict-resolution mechanisms, minimal recognition of rights to organise, and nested structures. They have been empirically linked to robust governance systems (Dietz et al. 2003, Ostrom 2005). Similarly, different governance principles have been proposed to support the design and analysis of various natural resource (co-)management arrangements operating at regional and state scales (Dovers 2001a, Lebel et al. 2006, Bellamy 2007, Lockwood et al. 2008). The list ranges from ‘value neutral’, such as resource allocation, level of formality, longevity, degree of autonomy (Dovers 2001a), up to ‘normative’, such as democratic representation, accountability and fairness (Lebel et al. 2006, Lockwood et al. 2008).

Application of institutional design principles in the analysis of governance arrangements has both benefits and challenges. On the one hand, principles offer general guidance regarding the core building blocks of institutional design required to structure specific management arrangements (Ostrom 1990, Dovers 2001a). To this end, they enable consideration of a wide range of conditions identified as ‘desirable’ in the design of management arrangements. On the other hand, the diversity of institutional settings and environmental problems requires a high level of generalisation (Young 2002). Therefore, some analysts acknowledge that the principles alone cannot determine whether a particular management arrangement will be effective in solving a problem (Young 2002, Ostrom 2005). Furthermore, as some authors note (Folke et al.}
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2007, Galaz et al. 2008), commonly identified conditions mostly describe social or economic aspects, ignoring other characteristics of environmental systems or resources these arrangements have been designed to address.

In response, an evolving trend is to approach the design and analysis of environmental governance sub-systems as a ‘problem of fit’ (Young 2002, 2008, Folke et al. 2007, Galaz et al. 2008). In essence, the concept of institutional fit builds on the assumption that the effectiveness of institutional design is a function of the fit between institutions and the biophysical and social domains in which they operate (Young 2002, Lebel et al. 2006, Folke et al. 2007). It builds on two fundamental propositions which can be briefly expressed as ‘no size fits all’ and ‘the same size does not fit all the time’. In other words, it is acknowledged that the regimes that perform well with one environmental problem can fail in other contexts (Young 2002). Similarly, those which performed successfully for a certain period of time may fail when environmental or social conditions change (Ostrom 2005).

The problem of fit is broad and can be approached from both social and ecological perspectives, and addressed at different levels of detail. As Folke et al. (2007:26) note, the list of issues of institutional fit ‘is almost infinite’. While the scope and application of the concept is still debated (Cox 2012, Vatn and Vedeld 2012), several analytical themes focusing on ecological linkages have evolved around the concept of scale (Cash et al. 2006, Cumming et al. 2006, Folke et al. 2007). According to Cumming et al. (2006:1), scale mismatches are situations ‘where the scale of environmental variation and the scale of the social organization responsible for management are aligned in such a way that one or more functions of the social-ecological system are disrupted, inefficiencies occur, and/or important components of the system are lost’. Currently, the literature distinguishes between spatial, temporal and functional scales as the core ‘environmental’ aspects of institutional fit (Cumming et al. 2006, Folke et al. 2007).

‘Spatial fit’, or alignment of institutional (i.e. management authority) and ecological (i.e. scales of ecosystem processes) boundaries, is one of the problems frequently emphasised in the resource management literature. As Cox (2012) summarises, ‘spatial fit’ contributes to internalisation of spatial externalities and allows the creation of a consistent system of rules which can be applied to the entire resource system. In the governance context, Cumming et al. (2006) distinguish between three potential types of spatial mismatches or misfits. The first is the case where fine-scale management units are holding control over the management of broad-scale environmental problems. In this case jurisdictions are unable to cope with the diversity of drivers affecting the maintenance of ecosystem processes (Cumming et al. 2006, Galaz et al. 2008). The second is the case where the broad-scale management units (e.g. state governments) are required to manage fine-scale ecological problems. This misfit can produce ‘one size fits all’ management regimes failing to account for important local variations (Cumming et al. 2006, Galaz et al. 2008). Finally, as many resource management systems are multi-layered in nature, the third
misfit can occur between ‘nested’ or hierarchical levels of ecological and social systems (Cumming et al. 2006).

‘Temporal fit’ involves the alignment of temporal scales of ecological and management processes. According to Cumming et al. (2006:3), temporal mismatches occur ‘when the temporal scales of management and the temporal scales of ecosystem processes do not align appropriately.’ Temporal fit tends to be discussed in several contexts. The most frequently emphasised misfits are short time horizons usually employed in policy-making and planning. It is acknowledged that short-term goals, which often are designed to suit electoral cycles, have created significant barriers for managing slowly changing ecological variables (Christensen et al. 1996, Folke et al. 2007). As a result, institutions produce decisions that assume a shorter time span than those embedded in the biophysical systems under consideration (Galaz et al. 2008).

The second misfit may occur in cases where the management system follows long term goals and fails to produce a timely response to fast changing ecological or social variables (Folke et al. 2007, Galaz et al. 2008, Young 2008). Finally, temporal misfits may also occur when management systems are designed too early or too late to cause desired ecosystem effects (Galaz et al. 2008).

Finally, ‘functional fit’ refers to the management scope or the variety of processes covered by particular governance sub-system. It concerns the extent to which the management scope matches ‘the nature, functionality, and dynamics of specific ecosystem it influences’ (Ekström & Young 2009:1). Functional misfits may occur when institutions lack provisions (i.e. do not regulate) for particular human interventions, or overlook key ecosystem links or side effects of the management (Young 2002, Folke et al. 2007). Misfits may also result from insufficient understanding of, or flawed assumptions about, ecosystem processes or dynamics. These may result in such practices as suppression of natural disturbances, or management models based on calculations of fixed ‘maximum sustainable yield’ or fixed ‘carrying capacity’ (Young 2002, Folke et al. 2007). Lastly, as separate management arrangements rarely cover the whole scope of interactions, functional misfits may also emerge as a failure to account for pressures created by other management regimes. In this case institutional misfits may emerge due to a lack of institutional mechanisms ‘to account for links within and among industry sectors and significant properties of the ecosystem’ (Ekström & Young 2009:1).

The concept of ‘fit’ has several implications for the design and analysis of environmental governance arrangements. First, it implies that there is no optimal governance model that should be applied to all environmental management problems. As Young (2008:20) points out governance arrangements should match ‘the defining features of the problems they address’.

Second, it does not offer a checklist of criteria or defining features. For example, while different scale mismatches or misfits are common, as Cumming et al. (2006) suggest they need to be considered only if they are producing or can potentially produce adverse ecological and/or social effects. Consequently, the design and analysis of environmental governance units requires
consideration of a variety of conditions both social and ecological that may operate as effectiveness determinants in specific settings. Third, as many scholars have emphasised, governance arrangements in the resource management field should be approached as experiments and changed in an adaptive fashion (Holling 1996, Dovers 1999, Ruhl 2005, Ostrom 2005). Therefore, as conditions change, different features may produce misfits over time.

Currently there is no consensus on the methodology for the assessment of institutional fit. However, a diagnostic approach has been suggested by Young (2002, 2008) and Ostrom (2009). How to approach the analysis of institutional fit is discussed in more detail in chapter 4.

3.5 Summary and conclusions

This chapter dealt with questions of what role institutions play in structuring human interactions with biodiversity, and what performance dimensions and criteria can characterise effective institutional performance. It examined the scope and characteristics of the biodiversity protection problem as described in the literature, particularly in academic sources. Placing biodiversity in the science, management and governance contexts, this chapter examined a range of propositions, findings and debates from several clusters of the literature, which provided different insights into various aspects of the problem and its institutionalisation.

As conceptualised by the environmental sciences, biodiversity describes all life forms. It is pictured as consisting of three core groups of elements - structure, composition and function - interacting across a hierarchy of several levels of biological organisation. The scope of biodiversity protection requires a range of management goals which can be pursued in pristine, semi-modified, and modified environments, including urban areas. The complexity and encompassing nature of the problem implies that institutional designs need to specify the scope of the problem, and provide a sufficient level of guidance tailored to particular management interventions, relevant spatio-temporal scales and levels of biological organisation. Important determinants of institutional environmental performance may be the validity of underpinning assumptions regarding the functioning of ecological systems, the available level of knowledge and capacities of knowledge providers and users.

Protection of biodiversity is shaped by two value paradigms: non-utilitarian and utilitarian. As an ecosystem property, biodiversity is increasingly linked to the concept of ecosystem services or benefits humans derive from various functions of the ecological systems. Identification and description of various groups of ecosystem services serves two purposes: to improve understanding of the role and value of biodiversity, and to support utilitarian market-based approach to conservation. As the studies suggest, the maintenance of ecosystem services and biodiversity protection are not perfectly correlated goals. Therefore, currently the achievement of
biodiversity protection outcomes requires institutional designs incorporating both utilitarian and non-utilitarian approaches.

Since the 1990s natural resource management policies and decision-making processes have been influenced by ecosystem management. As a new management paradigm it offers new ways of thinking about the functioning of the natural environment and how it responds to human disturbances. Ecosystems as management units are described as complex, dynamic systems interacting at various temporal and spatial scales, with uncertainty and surprise as inherent parts of their behaviour. Management has to consider various linkages between ecosystem components, ecologically relevant spatial and temporal scales, as well as to integrate multiple values and interests.

Uncertainty and limited knowledge is an inherent part of ecosystem management. Adaptive management as a structured decision-making process that has been widely promoted as a major management response to complex environmental problems, including biodiversity. It has been designed to integrate scientific knowledge in the decision-making process, facilitate collective learning and reduce uncertainty through research, modelling and monitored decision-making. However, despite widespread political appeal its ‘institutionalisation’ has encountered significant barriers. In practice there is no single set of rules that structures operational decision-making, and a large diversity of management practices have been promoted under the ‘adaptive management’ umbrella.

The ecosystem management literature contains numerous findings and propositions regarding how to approach the problem or a particular attribute. However, the institutional dimension has been insufficiently addressed. To clarify institutional determinants of ecosystem management, this study distinguishes between five broad institutional dimensions that, at two levels of the rules hierarchy, structure the implementation of ecosystem management. Important determinants are allocated authority boundaries and levels of discretion, as well as rules authorising and structuring the participation of new actors, such as knowledge holders in the policy-making processes. At the government level, ecosystem management can be enabled or hindered by the rules regulating funding allocation, as well as rules structuring and providing incentives for cross-sectoral cooperation and coordination and information exchange.

Environmental governance is another evolving concept which is applied to describe diverse structures and power relations in the management of environmental resources. During the last two decades the design and operation of local environmental governance has been shaped by several trends. The shift from ‘government’ to ‘governance’ has been characterised by decentralisation of decision-making power, and wide distribution of decision-making responsibilities among governmental, public and private actors. The literature distinguishes between several modes and strategies used as alternatives to a centralised top-down approach. Some of them have been promoted as ‘the solution’ to common-pool resource management
problems. However, accumulating evidence suggests that local environmental governance is a complex, hybrid system consisting of differently constituted actors interacting across several governance levels, each holding partial responsibility for the resolution (or creation) of environmental problems.

In the environmental governance literature institutions are approached from two perspectives: as an external system constituting governance sub-units, and as an internal system determining their capacity to manage environmental resources within the scope of authority. Several sets of principles of institutional design have been proposed to guide the constitution of specific governance arrangements. However, it is also recognised that there are ‘no recipes’ for how to design effective systems of environmental governance. The complexity of both social and ecological systems does not allow identifying general sets of criteria that would determine the effectiveness of particular arrangements. Therefore, the concept of ‘institutional fit’ has been suggested as an alternative approach to the design and analysis of different governance arrangements.

In conclusion, institutions perform a wide range of functions determining environmental outcomes. They range from creating an understanding of the environmental problem, coordinating and structuring specific management interventions, shaping capacities and motivations of involved actors, up to the framing of roles and power relations in the governance structures. Each aspect of institutional design can influence the ways in which the biodiversity protection problem is approached in specific contexts. Therefore, they all need to be examined in institutional analysis. At the same time, there are no specific sets of criteria that could serve as benchmarks or indicators for effective institutional design across different attributes, problems and contexts. Any specific proposition, including adaptive management, needs to be examined in the context of its underpinning assumptions, as well as enabling and constraining conditions. How this problem can be approached in the evaluation of institutional environmental performance, and what are the implications for assessment design is discussed in chapter 4.
Chapter 4 - Assessment of Institutional Environmental Performance: Diagnostic Framework

4.1 Introduction

This chapter completes the first part of this study exploring the theory and practice of institutional analysis and environmental performance assessment. Chapter 2 defined the effectiveness of institutional environmental performance, placing emphasis on institution – environmental problem causality. Chapter 3 explored biodiversity protection as a science, management and governance problem, and identified the broad range of functions institutions perform in structuring human-environment interactions, and the large variety of problem attributes that need to be considered in institutional design.

The complexity of institutional designs, diversity of regulated interactions and their contexts highlight two analytical questions: how to examine diverse determinants of institutional environmental performance in a systematic way? and, how to perform the analysis of diverse institutional designs?

As examined in chapter 3, the capacity of social systems to manage complex environmental problems such as biodiversity loss has been discussed in the context of various problem attributes such as uncertainty, dynamics and limited knowledge, connectivity linkages, knowledge integration, values and power relations. The institutional dimension, therefore, has been approached from different perspectives, and different sets of rules have been distinguished as opportunities or barriers for the sustainable management of ecological systems and biodiversity. However, no systematic approach has been developed to link a large variety of findings and recommendations to assist the analysis of multiple dimensions of institutional environmental performance.

This chapter addresses this gap and consists of two parts. The first part (section 4.2) introduces an approach for the design of a framework that can be applied to examine the performance of diverse institutional systems. Building on this foundation, the second part (section 4.3) proposes a systematic descriptive framework, which can be employed to examine multiple dimensions of institutional environmental performance, and discusses its application, limitations and potential.

4.2 Setting the foundation: institutional diagnostics and the Institutional Analysis and Development framework

This section consists of three parts. The first part introduces the ‘institutional diagnostics’ approach proposed by Young (2002) and outlines its application potential. The second part examines the typology of rules of the Institutional Analysis and Development (IAD) framework (Ostrom & Crawford 2005) as an approach for unpacking complex institutional designs. The
third part discusses how institutional diagnostics combined with the typology of rules can create the structure for a diagnostic framework.

4.2.1 Introduction to institutional diagnostics

Institutional theorists and analysts have encountered significant challenges in developing a set of principles for institutional design and assessment. As an alternative, several analysts have proposed a diagnostic approach (Young 2002, 2008, Ostrom 2007, 2009). In his book exploring the institutional dimension of environmental change Young (2002) has proposed ‘institutional diagnostics’ as a possible approach to the design of environmental regimes. Drawing on the analogy with medicine, the approach identifies a range of variables which can potentially affect the outcomes and tests whether they operate as causal determinants of particular problem (Young 2002). In essence, diagnostic analysis is closely linked to the concept of ‘institutional fit’ (see section 3.4.2), which requires the examination of problem attributes to determine whether institutions address or ‘match’ those attributes (Cox 2012).

According to Young (2002:176) the defining feature of institutional diagnostics is ‘an effort to identify important features of issues arising from environmental changes that can be understood as diagnostic conditions, coupled with an analysis of the design implications of each of these conditions’. Instead of attempting to classify the problem as a particular type and applying standardised ‘treatments’, institutional diagnostics seeks a disaggregation of the problem into the elements having relevance from a problem-solving perspective ‘and reaching conclusions about design features necessary to address each element’ (Young 2002:176). Young (2002) argues that institutional diagnostics offers two major benefits. First, it avoids problems of excessive generalisation required to establish principles of institutional design. Second, it enables identification of combinations of conditions having similar implications for institutional design. Consequently, as an exercise of ‘midrange generalisations’ institutional diagnostics avoids the limitations of treating each environmental problem as unique (Young 2002:176).

Young (2002) distinguishes between simple and complex forms of institutional diagnostics. By ‘simple’ diagnostics he describes the process in which the attributes of a specific environmental problem are examined separately (see Table 2 as an illustration). This approach builds ‘on a set of three linked procedures: identifying a range of diagnostic conditions, evaluating the design implications associated with each condition, and developing the interpretative skills necessary to apply these practices to specific cases’ (Young 2002:177). In contrast, ‘complex’ diagnostics involves consideration of interactions among two or several problem attributes. It implies that interactions between several components of a particular category of attributes, or between several components of different categories, may require different institutional solutions (Young 2002).
Both forms of diagnostic exercise have their strengths and weaknesses. While the simple diagnostics procedure is relatively easy to apply, it may not be sufficient to address complex management problems. As Young (2002) acknowledges, the major weakness of this form is an implicit assumption of no significant interactions between the elements of the problem. As a result, simple diagnostics may produce too simplistic or even inappropriate results. Complex diagnostics assist to overcome this challenge. Its major pitfall, however, is that consideration of a large range of interactions among various diagnostic conditions may lead to highly complex and convoluted assessment structures. Therefore, depending on the resources and capacities of the analysts, an application of a mixed strategy may be required (Young 2002).

Diagnostic analysis is not new. The importance of problem diagnostics as a crucial step in the design of social policies and programs is widely acknowledged and supported by methodological guidelines (Rossi et al. 2004). At the same time, institutional diagnostics has not yet evolved into a systematic process guiding the design and assessment of institutional environmental performance. There is no agreed-upon structure of problem elements that could assist with organising findings (Young 2002). Relatively recently, progress has been made in the analysis of small-scale social-ecological systems (SES). Ostrom (2007, 2009) has proposed a general framework consisting of seven sub-groups of variables affecting the self-organisation and robustness of common-property regimes.

At the current stage, an application of institutional diagnostics encounters several challenges. First, a lack of accumulated empirical knowledge implies that the application of the approach requires a substantial level of judgement. What classes of conditions can be distinguished, what and how many attributes form part of each class, and what implications they have for institutional design will largely depend on the judgement of the institutional designer or analyst. Second, as Young (2002:189) points out, there are no formulas for conducting the diagnostic
exercise and ‘no surefire way to avoid all diagnostic errors’. The analysis of institutional environmental performance requires consideration of both social and ecological factors. Therefore, the range of diagnostic errors can be significant. The analysts may encounter such problems as missing ecological data, flawed models or analytic constructs, as well as unidentified interactions or functional interdependencies (Young 2002). Problems with the selection of appropriate behavioural models may also emerge.

The institutional dimension poses another problem which remains unresolved in the proposed approach. Institutional diagnostics, as described by Young (2002), predominately focusses on the problem side of the equation, which is the disaggregation of the environmental problem into component parts (see Table 2 above). The institutional side, however, remains vaguely addressed. As Cox (2012) points out, the approach still lacks a typology of institutional arrangements that could underpin a systematic process of problem matching. In other words, the question remains of how to examine complex institutional systems and diagnose their ‘fit’. This problem is addressed in the next section.

4.2.2 Introduction to the IAD framework

The diversity of institutional systems presents an analytical problem of how to disaggregate the complex structures into the groups of features (performance dimensions) common across different designs. To address this issue the study proposes the application of the Institutional Analysis and Development (IAD) framework. The IAD framework has been developed by a group of scholars associated with the Workshop in Political Theory and Policy Analysis at Indiana University under the leadership of economist and institutional analyst Elinor Ostrom (Ostrom 1990, 2005, Ostrom et al. 1994). This framework and its potential for structuring the ‘diagnostics’ of institutional environmental performance is examined in this sub-section.

The IAD framework has been designed as a ‘multi-tier conceptual map’ to organise the inquiry into various aspects of institutional design and performance (Ostrom 2011:9). The primary value of the framework lies in a general set of variables or components, which Ostrom (2005) argues can be considered as universal building blocks crafting all regularised social interactions. These components can be identified at any governance level.

The focal level of the IAD framework is the so called ‘action arena’ which consists of an action situation and participants interacting in that situation (Ostrom 2005). Each action arena produces particular modes of interactions and outcomes which form the object of the analysis (see Figure 7). According to the framework, the structure of an action arena is shaped by three clusters of exogenous variables jointly affecting interactions and outcomes. The first consists of biophysical and material conditions which determine opportunities or constraints for possible actions. The

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10 Note: the recent review of the framework has simplified this component and replaced ‘action arena’ with ‘action situations’ without separating ‘participants’ and ‘situations’ (see Ostrom 2011).
second contains the attributes of the community within which the action arena is located. This cluster includes such attributes as accepted norms of behaviour, level of common understanding, homogeneity of preferences, composition, resource distribution and culture. The third cluster consists of the rules which provide ‘the set of instructions for creating an action situation in a particular environment’ (Ostrom 2005:17). Action arenas are not isolated and can be ‘nested’ in a hierarchical structure or linked horizontally, where the outcomes of one arena ‘become inputs into the next situation’ (Ostrom 2005:56).

Image has been removed.

**Figure 7** A framework for institutional analysis
Source: reprinted from Ostrom (2005:15)

Drawing on the game-theoretic approach, the IAD framework divides the structure of an action arena into seven elements that are regarded as an integral part of any action situation (see Figure 8). The first element is a set of participants (decision-making entities) which can be both individual and composite actors. The second deals with positions or ‘anonymous slots that are filled by participants’ (Ostrom 2005:193). Positions form a connecting link between participants and actions. The third element concerns potential outcomes that participants in positions can produce as a result of interactions. The fourth is a set of authorised actions that participants can undertake at a certain stage in the decision-making process. The fifth is the level of control allocated to particular positions over the choice of actions and outcomes (e.g. total control, partial control, no control). The sixth is the information available to participants about the decision-making situation. This includes information about available choices of actions, actions of other participants, outcomes, action - outcome linkages and the payoff structure. Finally, the seventh element consists of the external rewards or sanctions assigned to a particular course of actions or outcomes (see Ostrom 2005, Chapter 2 for a more detailed description of each element).

These elements form the basis for the horizontal classification of rules, which is the major variable of interest for this study. Rules defined as ‘shared understandings by participants about enforced prescriptions concerning what actions (or outcomes) are required, prohibited or permitted’ (Ostrom 2005:18) are external variables which determine an action arena. Depending
on which element is most directly affected (see Figure 8), the IAD framework distinguishes between seven groups of rules: position, boundary, scope, choice, aggregation, information and payoff (see Ostrom & Crawford 2005 for more detailed description of this classification).

Figure 8 Rules as exogenous variables affecting the elements of an action arena
Source: reprinted from Ostrom (2005:189)

According to this classification, position rules determine positions and the number of participants (i.e. institutional actors) in these positions. Boundary rules prescribe how institutional actors take up or leave positions. Scope rules define the overall scope of outcomes which the actors are undertaking to achieve. Choice rules specify actions that can be taken by an institutional actor holding a particular position. Aggregation rules determine how participants in positions collectively arrive at decisions, and what level of control each participant has over the final decision. Information rules specify content, sources, channels and frequency of information exchange. Finally, payoff rules determine costs and benefits assigned to choices of actions or outcomes in specific decision-making situations. As Ostrom (2011:19) argues, these groups ‘constitute the minimal but necessary set of rules needed to offer an explanation of actions and results’ within an action arena. A detailed description of this classification is provided in the final part of this chapter.

According to the IAD framework, the rules do not form isolated structures but are linked both horizontally and vertically. In addition to the horizontal classification, the IAD framework distinguishes between three hierarchical levels of rules described as ‘operational’, ‘collective-choice’ and ‘constitutional-choice’ rules, which cumulatively affect actions and outcomes at any setting. According to the vertical classification, operational rules guide day-to-day decision-making, including those directly affecting changes in the biophysical/material world. Collective-choice rules affect operational activities by determining ‘which participants, in what positions, chosen how, given information, and assessment of benefits and costs can make operational rules’
(Ostrom 2005:214). Finally, constitutional-choice rules determine participants, their mandates and other rules specifying development and change of collective-choice rules (Ostrom 2005).

Similarly, the vertical classification of rules of the IAD framework is generic. While described as ‘constitutional’, ‘collective choice’ (policy-making) and ‘operational’, these groups do not imply particular governance levels or types of involved actors. For example, constitutional-choice rules may operate within state government settings, as well as within the boundaries of contractual arrangements concluded between private individuals (Ostrom 2005). Furthermore, the hierarchy does not imply a distinct set of decision-making arenas. At each level, there may be several arenas producing particular groups of rules (i.e. institutional outputs), which further shape the interactions and outcomes of selected action arenas (Ostrom 2005).

Application of the IAD framework to complex institutional designs offers several benefits. First, while various frameworks have been offered in the literature, the IAD framework is one of the few that has been empirically tested across numerous cases (Ostrom et al. 1994). Several studies have proved that its application is not restricted to specific management arrangements (Margerum & Born 2000, Morrison et al. 2004, Koontz 2006, Basurto et al. 2010, Heikkila et al. 2011). Second, the IAD framework is generic and does not prescribe the application of particular methodologies or theoretical frameworks (Ostrom 2005, Koontz 2006). Thus, it provides a good platform for the analysis of different types of interactions and institutional settings. Finally, the classification of rules is flexible. The authors acknowledge that further structuring may be required for more complete specification of rules affecting particular components of an action situation (Ostrom & Crawford 2005). Consequently, generic categories can be further adapted, as well as expanded into the sub-sets of variables tailored to particular problems and contexts.

Application of the IAD framework for the analysis of institutional environmental performance has one major limitation. The framework explicitly recognises that rules are only one of the factors affecting an action situation. In contrast to the concept of ‘institutional fit’ (see section 3.4.2), the framework does not expand on linkages between external elements, and considers ‘rules’, ‘biophysical/material conditions’ and ‘attributes of the community’ as independent variables shaping the structure of ‘the game’ (see Figure 7 above). Consequently, to use it for analysing institutional environmental performance, the framework needs to be ‘animated’ by theories addressing linkages between social and ecological systems. How to adapt the typology of rules for the analysis of institutional environmental performance is discussed next.

### 4.2.3 Linking institutional diagnostics and the IAD framework

Institutional diagnostics as described by Young (2002) is a systematic process for identifying and structuring different problem attributes having implications for institutional design. The IAD framework allows unpacking complex institutional designs with nested hierarchies of rules into generic sets of performance dimensions. This study argues that combining both approaches can provide a sound foundation for the design of diagnostic frameworks for analysing a large
diversity of institutions. Therefore, the final part of this section describes three parts of the framework design process: selection of problem attributes, description of core categories of rules (performance dimensions), and identification of linkages between problem attributes and respective dimensions of institutional performance. The output of this process is the recommended diagnostic framework presented in the second part of this chapter.

**Problem categories and attributes**

The most complicated part in institutional diagnostics is selection of ‘diagnostic conditions’ (Young 2002) or problem attributes having implications for institutional design and performance. While analysts have identified a range of attributes that need to be considered in the design of environmental institutions (Dovers 1995, 2001a b, Young 2002, Mickwitz 2003), there is no unified approach for dealing with the structure of environmental problems. Systematisation of diagnostic conditions is further complicated by the diversity of environmental problems and their socioeconomic, institutional and biophysical contexts. As Young (2002:177) acknowledges:

There is little prospect that we can devise a typology of diagnostic conditions that is exhaustive. As in most diagnostic endeavours, the possibility of encountering conditions associated with specific environmental problems that are unfamiliar but turn out to have important prescriptive implications is always present.

Despite these challenges, the accumulated body of knowledge allows distinguishing a range of attributes of environmental problems. Drawing on Young (2002), this study proposes to group them into three categories: attributes of ecological systems, attributes of management interventions and attributes of institutional actors. As outlined in chapter 3 (see section 3.3.1), ecosystem management identifies a range of attributes of ecological systems having implications for management. Multiple spatial and temporal scales, connectivity, resilience, uncertainty and dynamics need to be considered in institutional design and analysis. They are the major determinants of ‘institution-ecosystem fit’ or institutional capacity to produce intended environmental outcomes. Attributes of management interventions or human pressures are a closely related category. However, it contains a range of independent attributes: intensity, timing, longevity, connectivity, magnitude and predictability of impacts, which have separate implications for institutional designs (Dovers 1995). Finally, actor attributes are characteristics of the group of actors involved in causing or resolving environmental problems. As discussed in chapter 2 (see section 2.2), institutional theorists build their models based on three groups of behavioural attributes: ability to capture the problem, capacity to take actions, and motivation or preferences to do so. In the environmental problem context, the group of actors is not homogenous. At least three sub-categories can be distinguished in the management ‘arena’: direct beneficiaries (e.g. resource appropriators, users, polluters), affected stakeholders, and management bodies performing different governance functions.
Table 3 on the next page categorises the proposed three classes of diagnostic conditions into several groups of problem attributes that have implications for institutional designs across a variety of environmental problems.\textsuperscript{11} It draws on the sources reviewed in the previous two chapters, as well as on the work of analysts exploring the structure of environmental problems, particularly Dovers (1995, 2001a b), Young (2002, 2008) and Ostrom (2005, 2007, 2009). The table contains two components. The first is a list of key problem attributes grouped according to the three categories outlined above. The second component is the potential characteristics of each attribute which, drawing on Dovers (1995), are described in terms of gradients and scales. This list incorporates some of the factors that need to be considered in institutional assessment, as different characteristics and gradients may require different regulatory responses. This list, however, does not imply a particular measurement system and, as Dovers (1995:99) puts it, includes characteristics that are rather ‘gradients for approximation on the basis of informed judgement’.

\textit{Environmental performance dimensions}

The concepts of ‘institutional fit’ and ‘social-ecological systems’ place emphasis on interdependencies and treat institutions as an interface between the biophysical and social systems (see section 3.4.2). To some extent, these concepts blur the boundaries by considering humans and ecosystems as two interacting groups of actors ‘in the arena’. Humans derive benefits from goods and services of ecological systems or their destruction, and attempt to understand (research and monitor), guess (model), as well as ignore past, present or future response of affected systems. Consequently, institutions structuring human-environment interactions can be seen as explicitly or implicitly built on assumptions about the behaviour of human actors and assumptions about the state or behaviour of ecosystems or their properties.

\textsuperscript{11} Note: while this set of attributes provides a starting point for institutional diagnostics, specific environmental problems may require distinguishing additional problem attributes (e.g. ‘productivity of the system’ in extractive resource management).
### Table 3 Problem attributes affecting institutional environmental performance

<table>
<thead>
<tr>
<th>Attributes of ecological systems or their properties</th>
<th>Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Location and spatial extent</td>
<td>Within boundaries of a single jurisdiction – crossing few jurisdictions - crossing multiple jurisdictions (e.g. national, global)</td>
</tr>
<tr>
<td>Temporal scale</td>
<td>Fast (months, years) - medium (years, decades) – long term (decades, centuries)</td>
</tr>
<tr>
<td>(e.g. recovery, response)</td>
<td>Relatively simple homogeneous systems – complex systems with a large number of elements playing functionally important roles</td>
</tr>
<tr>
<td>Complexity and interdependence</td>
<td>Contained within jurisdictional boundaries – affecting other jurisdictions (e.g. habitats for migratory species)</td>
</tr>
<tr>
<td>Connectivity linkages</td>
<td>Sufficiently stable and predictable – unpredictable with rapid shifts and non-linear change</td>
</tr>
<tr>
<td>Systems dynamics</td>
<td>Resilient to certain levels or types of disturbances – vulnerable – highly vulnerable (e.g. tipping points have been reached)</td>
</tr>
<tr>
<td>Capacity to absorb disturbances or functional limits</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Attributes of management interventions (environmental impacts)</th>
<th>Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Location and spatial extent</td>
<td>Within boundaries of a single jurisdiction – crossing few jurisdictions – crossing multiple jurisdictions (e.g. national, global)</td>
</tr>
<tr>
<td>Timing of occurrence</td>
<td>Near (months, years) – medium (years, decades) – long term (decades, centuries)</td>
</tr>
<tr>
<td>Longevity of possible impacts</td>
<td>Ongoing – short (months, years) – medium (years, decades) – long term (decades, centuries)</td>
</tr>
<tr>
<td>Reversibility</td>
<td>Easily (inexpensive) or quickly (quick recovery) reversible – expensive/difficult – irreversible (permanent loss)</td>
</tr>
<tr>
<td>Connectivity of impacts</td>
<td>Low connectivity (contained within jurisdictional boundaries) – high connectivity (environmental externalities, cross-boundary impacts)</td>
</tr>
<tr>
<td>Cumulative impacts (pressures)</td>
<td>Single pressure – diversity of pressures produced by various management interventions</td>
</tr>
<tr>
<td>Magnitude of possible impacts</td>
<td>Low – moderate – severe – catastrophic</td>
</tr>
<tr>
<td>Predictability</td>
<td>The extent (including cumulative and cross-boundary) and nature of impact is well known – uncertain (risk) – not known (ignorance)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Actor attributes</th>
<th>Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Direct beneficiaries (appropriators/users/polluters)</td>
<td>Few – small number – large number of actors entitled to affect ecological systems or their properties</td>
</tr>
<tr>
<td>Number of participants (actors) producing environmental pressures</td>
<td>Restricted – moderately restricted – unrestricted rights to affect ecological systems or their properties</td>
</tr>
<tr>
<td>Level of control</td>
<td>Low capacity (e.g. insufficient resources, technology, low demand for produced goods) – moderate – high</td>
</tr>
<tr>
<td>Capacity to produce impacts</td>
<td>Good – sufficient - insufficient level of knowledge/information about affected systems/properties or the extent or nature of produced impacts</td>
</tr>
<tr>
<td>Level of knowledge, available information</td>
<td>High – sufficient - insufficient capacity to gain knowledge (learn) or obtain information (e.g. finance monitoring or purchase data)</td>
</tr>
<tr>
<td>Capacity to obtain knowledge, information</td>
<td>High interest (internal values+ external benefits) – moderate – low or no interest</td>
</tr>
<tr>
<td>Interest to sustain ecological systems or their properties</td>
<td>High – moderate – low – no capacity (e.g. lack of participation rights) and/or interest (e.g. no trust, lack of shared values, conflicting interests)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Stakeholders</th>
<th>Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number and location of affected stakeholders</td>
<td>Small – moderate - large number of stakeholders located within single – few – many jurisdictions (e.g. different states)</td>
</tr>
<tr>
<td>Affected interests/benefits</td>
<td>Low – moderate – high impact on interests/benefits of other stakeholders (e.g. impact on other resource sectors)</td>
</tr>
<tr>
<td>Homogeneity of interests</td>
<td>Common interests (e.g. shared benefits) in sustaining ecosystems or their properties – compatible interests – conflicting interests</td>
</tr>
<tr>
<td>Level of knowledge, available information</td>
<td>Good – sufficient - insufficient level of knowledge/information about affected systems/properties or the extent or nature of impacts</td>
</tr>
<tr>
<td>Capacity to obtain knowledge/information</td>
<td>High – sufficient - insufficient capacity to gain knowledge or obtain environmental information; information asymmetries</td>
</tr>
<tr>
<td>Capacity or interest to engage in policy- or collective decision-making</td>
<td>High – moderate - low - no capacity (e.g. lack of participation rights) and/or interest (e.g. no trust, lack of shared values, conflicting interests)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Management bodies</th>
<th>Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Authority level (legal mandate)</td>
<td>Good – sufficient – insufficient capacity to affect environmental outcomes (e.g. restrict users, monitor) or engage in cross-boundary interactions</td>
</tr>
<tr>
<td>Financial capacity</td>
<td>Good – sufficient – insufficient capacity to undertake required functions, obtain required environmental knowledge or information</td>
</tr>
<tr>
<td>Level of knowledge, available information</td>
<td>Good – sufficient – insufficient level of knowledge/information about affected systems/properties or the extent or nature of impacts</td>
</tr>
<tr>
<td>Motivation to protect or sustain ecosystems or their properties</td>
<td>High – moderate - low - no motivation to undertake actions or make decisions regarding protection of ecosystems or their properties</td>
</tr>
<tr>
<td>Representation of community of interests</td>
<td>Good – sufficient - insufficient representation of community benefiting from protection or sustained use of particular systems/properties</td>
</tr>
<tr>
<td>Accountability and decision-making transparency</td>
<td>High - moderate - low accountability or transparency - no obligation to provide information on decisions and outcomes</td>
</tr>
</tbody>
</table>

Viewing humans as an integral part of ecosystems, and ecosystems as separate actors or ‘service providers’ with their own attributes, requires rethinking the structure of ‘the game’.\textsuperscript{12} Therefore, to understand how different groups of rules contribute to the creation or resolution of diverse environmental problems, it is useful to modify the description of generic categories of the IAD framework to explicitly link them to different attributes of ecosystem, human and human-environment categories (see Table 3 above). Furthermore, such modification can also assist with scoping rules for the analysis of complex institutional systems, which may require distinguishing between environmental and non-environmental dimensions of institutional performance.

Table 4 provides a modified description of the categories of rules of the IAD framework to structure the diagnostics of institutional environmental performance. It maintains the initial categories and the rationale underpinning the typology of rules of the IAD framework (Crawford & Ostrom 1995 and Ostrom & Crawford 2005). The only adjustment introduced is the merging of two categories of rules, namely ‘position’ and ‘boundary’, into a single category called ‘position-boundary’. This adjustment is proposed for analytical purposes, because position rules, which specify positions and a number of participants in these positions, can be rarely analysed separately from boundary rules defining who is eligible to hold the position and under what conditions. Detailed description of these performance dimensions and their sub-categories will be provided in the final part of this chapter.

Based on the ‘institutional grammar’ introduced by Crawford and Ostrom (1995), Table 4 considers differences in the potential of the configurations of rules to produce uniform behavioural responses. The term ‘prescribe’ is applied to describe rules which require mandatory behaviour or actions that institutional actors ‘must’/‘should’ or ‘must not’/‘should not’ undertake. The term ‘permit’ is applied to describe the rules which enable or authorise institutional actors to take particular actions, choose among alternatives, or to decide whether any action should be taken (i.e. ‘may’/‘may not’ undertake). In the latter case actors hold a certain level of control over the choice of possible sets of actions and outcomes which can be affected by their perceptions or preferences.

\textsuperscript{12} Note: in this context the term ‘game’ is used broadly. This statement does not imply that changes need to be made in the game theory where ‘nature’ is not considered as a separate actor.
Table 4 Dimensions of institutional environmental performance

<table>
<thead>
<tr>
<th>Category</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Scope rules</strong></td>
<td>Specify what environmental outcomes (scope of behavioural change) and/or impacts (scope of biophysical change) as a result of what interactions in what geographical area(s) and within what timeframe(s) involved institutional actor(s) in position(s) are prescribed or permitted to achieve or prevent</td>
</tr>
<tr>
<td>Position-boundary rules</td>
<td>Specify which and how many positions having authority to produce or affect environmental outcomes and/or impacts can be established and when, how and under what conditions participants are prescribed or permitted to enter or leave these positions</td>
</tr>
<tr>
<td><strong>Choice rules</strong></td>
<td>Specify the scope of actions that institutional actor(s) in position(s) are prescribed, permitted or prohibited to undertake in the light of specified conditions to produce or prevent particular environmental outcomes and/or impacts</td>
</tr>
<tr>
<td>Aggregation rules</td>
<td>Specify which institutional actors in positions are involved in joint decision-making regarding the achievement or prevention of particular environmental outcomes and/or impacts, how decisions are made and what level of control each actor has over the final decision</td>
</tr>
<tr>
<td><strong>Information rules</strong></td>
<td>Specify what kind of environmental information in what form, in what timeframes, from what sources and through what channels institutional actors in positions are prescribed or permitted to obtain, apply or to provide to other actors</td>
</tr>
<tr>
<td>Payoff rules</td>
<td>Specify what incentives (benefits) or disincentives (costs) institutional actors in positions must or may face with regard to the choice of actions producing or preventing particular environmental outcomes or impacts or achieving particular level(s) of environmental performance</td>
</tr>
</tbody>
</table>

Source: author based on Ostrom and Crawford (2005)

Problems with institutional environmental performance rarely revolve around a single action situation or a single level of the rules hierarchy. The ecosystem management and related literature frequently addresses two levels of interactions or management arenas (see section 3.3.3). For example, such elements as participation of stakeholders, scientists and environmental knowledge holders play significant role at the policy-making (i.e. collective-choice) level. Consequently, in many situations the analysis of the institutional dimension needs to consider two interacting levels of rules cumulatively affecting ‘on ground’ outcomes.

As demonstrated in the final part of this chapter (see Tables 5 and 6 at the end of section 4.3.1) the proposed categories can be further adjusted to distinguish between the vertical levels of rules hierarchy. For example, the group of scope rules may include the rules describing what environmental outcomes or impacts institutional actors are expected to produce within a particular area, as well as rules specifying what scope, scale and type of interactions the involved actors are required or permitted to regulate. As a result, operational rules can be explored as an output variable for the analysis of collective choice rules, and as a causal variable for ‘on ground’ environmental outcomes or impacts.
Linking problem attributes and performance dimensions

The design of the diagnostic framework implies the need to identify how particular problem attributes can be affected by institutions, which performance dimensions play a crucial role, and what are the ‘desired characteristics’ of their design. Thus, the design of the framework can be described as a two-step process. The first involves linking selected problem attributes to their institutional determinants. The second requires the identification of desired characteristics of respective performance dimensions, and these can be translated into a set of diagnostic questions.

The first step of the framework design involves specifying which performance dimensions of the IAD framework most directly affect a particular problem variable (i.e. potentially contribute to institutional ‘fits’ or ‘misfits’). For example, a frequently emphasised requirement for the involvement of scientists and other knowledge holders in the rule setting process can be linked to the design of position-boundary rules (collective-choice level) authorising such participation, as well as aggregation rules determining the role of these actors in the joint decision-making process. Motivation of institutional actors to consider the protection of ecological systems can be directly linked to the cost-benefit or rewards-sanctions structure established by the payoff rules. This exercise may also require consideration of alternative groups of rules. For example, problems with spatial ‘fit’ or mismatch between jurisdictional and ecological boundaries can be linked to the design of scope rules defining the overall problem domain or, alternatively, to the design of mandates (i.e. choice rules) requiring or enabling institutional actors to consider cross-boundary aspects. The main output of this step is a list of various dimensions of institutional environmental performance distinguished under different categories of rules of the IAD framework.

The second step involves formulation of a set of diagnostic questions. As noted in chapter 3, environmental and social scientists have provided a range of characteristics of institutional design features ‘fitting’ well or poorly with particular problem attributes. For example, the problem of ‘temporal fit’ may require examining whether management timeframes incorporated in scope rules ‘match’ ecological scales of the system or its properties (e.g. lag effects in ecosystem response, life cycles of species). Problems linked to ecosystem connectivity can trigger questions whether the rules enable consideration of cross-boundary effects (choice rules), or require or enable provision of environmental information to other institutional actors about intended environmental impacts (information rules). Diagnostic questions, thus, present an opportunity to cover a large variety of factors which can potentially affect institutional environmental performance. They can assist the analyst with identifying which institutional variables operate as main determinants of environmental outcomes or impacts in specific settings.
The design of the framework also raises questions regarding the level of detail which should guide the diagnostic exercise. The literature contains numerous specific recommendations on management and policy responses which can be linked to particular problem attributes (see Table 1 in section 3.3.3). Similarly, Young (2002) demonstrates that a simple diagnostic approach could be applied to identify specific implications for institutional design (see Table 2 in section 4.2.1). At this stage of the research, however, the possibility of identifying particular sets of general ‘treatments’ linked to separate problem attributes is questionable. In this context, a good example is adaptive management, which on the theoretical level has been advocated as almost a universal solution to problems characterised by uncertainty and limited knowledge. In practice, due to a range of other constraining conditions, the approach has been implemented in only a limited number of cases (see section 3.3.2). Similarly, promotion of a particular institutional actor as the ‘best’ regulatory authority, drawing on few attributes such as accountability, democratic representation and ‘closeness to ecosystems and people’, has turned out to be insufficient guidance for the design of governance systems (see section 3.4.1).

The above problems suggest, that instead of attempts to ‘diagnose’ whether an established institution supports specific management/governance response strategies, the analysis needs to take a ‘performance-based’ orientation accepting the diversity of potential solutions. This, however, does not exclude the possibility to add more specific sets of criteria when examining particular institutional arrangements. A detailed description of environmental performance dimensions and the diagnostic framework is presented and discussed in the final part of this chapter.

4.3 Environmental performance assessment: diagnostic framework

This section completes this chapter and the first part of this study by proposing the diagnostic framework that can be employed to structure the analysis of institutional environmental performance. This section consists of two parts. The first part presents the framework and describes its core groups of rules. The second part discusses its benefits, limitations and application potential.

4.3.1 Diagnostic framework: categories of rules

This sub-section describes the categories of rules introduced in the previous part of this chapter, as well as outlines how a particular group of rules relates to different aspects of ecosystem management and biodiversity protection problems examined in previous chapters. A detailed description of the typology, related dimensions of environmental performance, and guiding diagnostic questions are presented in Tables 5 and 6 at the end of this sub-section. Table 5 provides the framework for the analysis of rules guiding interactions at the operational level, or the level most directly affecting ‘on ground’ outcomes and impacts. Table 6 addresses so called
collective-choice rules, or rules shaping interactions of institutional actors involved in the
design and change of operational rules.

**Scope rules**

According to Ostrom and Crawford (2005) scope rules define outcome variables. In the
environmental performance context they specify the problem domain. Scope rules determine
several boundaries of the problem domain, such as spatial or geographical scale (e.g. river basin,
local government area), regulated ecosystem properties (e.g. fish species, vegetation, water), as
well as types of management interventions (e.g. development of protected areas, vegetation
clearing, pollution). The rules can also establish temporal boundaries by specifying time periods
for the achievement of expected outcomes or impacts (e.g. timeline for phasing out polluting
sub-stances). In the case of multiple or conflicting goals the rules may also define the scope of
environmental performance by specifying how the goals should be prioritised. On the rule
setting (collective choice) level, scope rules define the overall scope and scale of the
environmental problem that the involved institutional actors are authorised to resolve through
the design or change of operational rules.

Scope rules can be structured and formulated in a variety of ways. They can be expressed as
expected states of ecological systems, or their properties, or expected levels of performance
(e.g. pollution or resource extraction level) an institution has been designed to achieve. In
complex institutional systems the overall management scope can be sub-divided into several
problem domains specifying expected outcomes or impacts for particular interactions,
ecosystem properties or locations. The rules can be stated in terms of specific environmental
outcomes or impacts, or contain broad normative statements (Slocombe 1998b for ecosystem
management goals).

Scope rules define the overall potential of established institutional systems to address
environmental problems underpinning their designs. The analysis of these rules can be linked to
several dimensions of ‘institution – ecosystem fit’ (see section 3.4.2). For example, problems of
’spatial fit’ may emerge in cases where delineated jurisdictional boundaries do not match
geographical areas required for the regulation. ‘Functional misfits’ may be produced when the
scope of interventions leaves unregulated gaps (Ekström & Young 2009). Misfits may also
occur if specified performance levels are based on flawed assumptions about the functioning of
ecological systems or their response to human disturbances. Furthermore, as several institutions
can be designed to deal with different aspects of connected ecological systems, problems can be
created if defined levels of performance fail to consider cumulative pressures. Finally, ‘temporal
misfits’ may occur when incorporated temporal scales do not reflect the scales of affected
ecosystem processes or are too short or too long to respond to environmental change
(Christensen et al. 1996, Young 2002).
Position-boundary rules

Position-boundary rules comprise design features specifying which actors (both collective and individual) are prescribed or entitled to participate in the environmental decision-making (operational level) or rule setting process (collective choice level). The rules specify what and how many positions are established, which participants are eligible, and how they enter or leave the positions (Ostrom & Crawford 2005). According to Ostrom and Crawford (2005) there are several ways in which the rules can determine position holders. First, the rules can regulate position uptake through eligibility (boundary) requirements by prescribing required personal or acquired attributes of potential position holders. For example, to enter a position of environmental expert, participants may be required to meet such eligibility criteria as relevant education or experience. Access to the resource (i.e. position of resource user) could be made available to those participants who, for example, hold the licence, property rights or possess particular technology. Second, position uptake can be regulated through the rules determining how participants enter positions. According to Ostrom and Crawford (2005) the participant may have either a full control over the decision, or the rules may require nomination or authorisation from other position holders. In some cases there can be a compulsory requirement to hold a position. Finally, the rules can determine how long the participant can hold a position, and whether and under what conditions the participant is required or allowed to leave.

To understand the role each position plays in the management of environmental problems, position-boundary rules need to be explored together with the choice rules specifying the scope of actions allocated to particular positions (see below). However, there are several ways the design of position-boundary rules can affect environmental outcomes and impacts. First, on the appropriator/polluter/user level the rules may determine the number of actors which can enter positions linked to the resource use rights (e.g. consumptive use, pollution, land use transformation) and consequently affect cumulative pressures on the ecological systems or their properties. Position-boundary rules may also affect the number of actors by specifying particular attributes or conditions restricting or enabling uptake of these positions (see Ostrom 2005 for different examples). Furthermore, environmental outcomes may be affected through inclusion or exclusion of certain participants. For example, studies of small-scale CPR institutions have revealed that resource use outcomes can differ depending on whether the use rights are allocated to participants having a stake in the community, or to external appropriators who lack a long term interest in the sustainability (Ostrom 2005).

Second, at the management level, position-boundary rules determine the structure of environmental decision-making. Power relations, allocation of management (administrative) positions and public and stakeholder participation in the decision- and policy-making processes are the core topics of the environmental management and governance literature (see sections 3.3 and 3.4). There is a growing consensus that participation of non-governmental actors, such as
indigenous and local communities, resource users and other stakeholders, forms an integral part (i.e. require positions) of environmental governance. Of particular importance for environmental outcomes is the involvement of participants holding environmental knowledge, as well as participants having interests in sustaining environmental values. ‘Misfits’ can be produced when the allocation of management positions occurs without matching the skills, capacities and interests of potential participants to those required to undertake the scope of allocated actions (see SCBD 2004 for a set of diagnostic criteria). In this context the most frequently mentioned governance problem is the operation of funding rules which largely determine the capacity of the regulatory authorities. Environmental outcomes can be also affected if the number of actors entitled to hold a particular position is not sufficient for carrying out allocated functions, or the required position is not established.

**Choice rules**

Choice rules determine the scope of authority of each position-holder. The rules specify what actions, and where and when institutional actors in positions, are required, permitted or forbidden to undertake under specified conditions (Ostrom & Crawford 2005). For example, the rules can permit a group of actors holding a ‘user’ position to interfere with the ecological systems if produced impacts do not exceed certain levels (e.g. maximum levels of pollutant release), or thresholds (e.g. minimum vegetated area to sustain the habitat), or they do not occur within certain locations (e.g. protected areas) or periods of time (e.g. drought season). The rules may be highly specific or may allocate to institutional actors a high level of discretion to choose among available alternatives. As Ostrom and Crawford (2005) note, the regulators may choose between different combinations of choice and scope rules. Choice rules may be designed to describe a set of actions which institutional actors must/must not/may undertake or, alternatively, the scope rules may prescribe the required level of performance allowing institutional actors to decide how to achieve that level. On the rule setting (collective-choice) level, choice rules specify what configurations of operational rules under what conditions involved institutional actors are required or permitted to develop or change.

Choice rules are the core group of rules determining the extent of environmental impacts, which can be produced under a particular institutional design. On the user/polluter/appropriator level choice rules determine the rights and duties of eligible actors. They may range from highly restricted and regulated rights of intervention to almost unrestricted rights enabling actors ‘to do what they decide’. Allocated rights can also be balanced by a set of duties to maintain a certain state of the environment (e.g. management of introduced species, retention of riparian vegetation) or to compensate for environmental externalities. On the management level, the allocated scope of authority determines the capabilities of the management bodies to coordinate,

13 Note: this approach to regulation is often referred to as performance-based approach (see Coliagnese et al. 2003).
enable, or restrict actions producing environmental pressures and monitor compliance. Similarly, the scope may range from advisory or coordinative functions up to the level which enables the regulatory authorities to constrain actions of resource users and enforce compliance.

There is a variety of ways the design of choice rules can affect institutional environmental performance. In general, institutions regulating human-environment interactions are explicitly or implicitly built on a set of assumptions about the state of ecosystems and their response to regulated disturbances. Therefore, performance problems can occur in situations where the rules do not make ‘ecological sense’ (Galaz et al. 2008), or where prescribed or permitted actions, their scope, timing or location ignore or are based on flawed assumptions regarding the characteristics of affected ecological systems or their properties (e.g. natural disturbance regimes, productivity, life cycles or habitat ranges of species). ‘Functional misfits’ may also occur in cases where institutional designers fail to consider impacts of other disturbances, or impacts on connected properties or systems. In this context, problems of spatial and functional fit can be linked to decision-making fragmentation and to the design of mandates which preclude cross-boundary or cross-sectoral interactions. An important determinant of environmental outcomes may also be the level of discretion available to institutional actors to tailor their decisions to ecosystem- and location-specific conditions, acquired environmental knowledge, and/or observed changes in the biophysical environment.

**Aggregation rules**

Aggregation rules specify which institutional actors in positions are required or permitted to be involved in joint decision-making, how decisions are made, and what level of control each actor has over the final decision. According to Ostrom and Crawford (2005), aggregation rules are a necessary part of institutional design whenever multiple actors in positions hold partial control over the same set of action variables. They distinguish between three generic forms of aggregation rules: non-symmetric aggregation, symmetric aggregation, and lack of agreement rules. In non-symmetric situations, a single actor or a small group may hold control over the final decision. In symmetric aggregation institutional actors may have joint control and may apply unanimity, consensus or majority voting rules to reach the decision. In ‘lack of agreement cases’ aggregation rules may specify the outcome such as continuation of the status quo, or determine which actor is entitled to resolve the problem (Ostrom & Crawford 2005).

A common governance problem in the management of complex ecological systems is decision-making fragmentation, where various decision-making entities hold partial control over the activities impacting upon the environment. By specifying which institutional actors are required or enabled to participate in joint decision-making, aggregation rules can affect the overall problem domain and the scope and capacity of joint decision-making authority (see e.g. Williams et al. 2009 on the role of these rules in adaptive management projects). Consequently, by linking key positions in the decision- or policy-making process, aggregation rules can
operate as important determinants of spatial and functional ‘fits’ and ‘misfits’. These rules can also affect the outcomes by specifying the decision-making process and determining ‘how much weight each participant will have relative to others’ (Ostrom & Crawford 2005:202). As Ostrom (2005) points out, the level of control particular actors can have over the decision-making process or outputs may determine whose interests will be taken into account.

The design of collective decision-making processes has been widely discussed in the context of ecosystem management and environmental governance. Stakeholder involvement and their role in policy-making process is an important ‘ingredient’ of institutional design. Moreover, as environmental decision-making tends to be conflict driven (Lee 1993) established decision rules can indicate whether, and to what extent, involved actors should consider environmental interests. For example, different environmental outcomes and outputs (e.g. operational rules) can be produced when control over the final decision is held by the agency holding an environmental protection mandate (i.e. institutional role), or by an agency undertaking to promote development and economic growth. As Young (2002) has demonstrated decision rules, such as allocated veto rights in combination with established payoff structures, may create ‘institutional gridlock’ and prevent a change of actions even if adverse environmental impacts are obvious.

**Information rules**

Information rules specify what environmental information, and from what sources, institutional actors in positions are required or permitted to obtain, apply or provide. These rules determine the level of information about the overall structure of the situation, the state of the system variables, previous actions, as well as previous and current actions of other actors (Ostrom & Crawford 2005). While the IAD framework primarily focuses on rules determining information exchange (Ostrom & Crawford 2005), this study distinguishes between three groups of information rules that determine institutional environmental performance.

The first group regulates environmental information that institutional actors obtain or provide to evaluate their progress towards the achievement of specified environmental goals or objectives (e.g. environmental criteria, indicators). The second group specifies environmental information on the choice of the actions or impacts (e.g. environmental impact statements, performance reports) that institutional actors are required to obtain or to provide to others, including the general public. Both groups may also specify the content (e.g. scope of data), authorised sources (e.g. expert advice, local knowledge, data base), form or timing of required information, as well as the information gathering process. The third group of rules specifies what environmental information, how often, in what form, and through what channels, institutional actors are prescribed or permitted to exchange. In other words, information rules comprise a range of dimensions providing substantial guidance, and prescribing or enabling information generation or exchange.
Information rules are regarded as one of the main determinants of the capacity of institutional actors to understand the decision-making situation. Complex environmental problems such as biodiversity protection require ‘institutionalisation’ of the environmental knowledge. Whether expressed as management standards, guiding principles, lists of protected species, or indicators defining the levels of resource extraction, this information guides, enables or restricts management actions. As most environmental decision-making is carried out under conditions of uncertainty, established information rules form an important part of the ‘adaptive capacity’ (Folke et al. 2002) of actors to learn and adjust management responses. Furthermore, as decision-making authority tends to be distributed among a large number of positions, information rules affect the capacity of institutional actors to consider cumulative or cross-boundary effects.

Integration of environmental information in decision- and policy-making processes is a dominating theme in the environmental sciences (see section 3.2.3) and ecosystem management literature (see Table 1 in section 3.3.3). In general, the availability of environmental information depends on factors such as the complexity of the problem, existing levels of knowledge, as well as the level of predictability of the system’s functioning and response (Christensen et al. 1996). While the rules alone cannot determine what choices will be made, problems may be caused if they fail to reflect the management problem (e.g. if performance criteria or indicators are not relevant), or are based on flawed assumptions regarding the capacities of the decision-makers to obtain or understand the required information, or of the knowledge holders to provide it within specified timeframes. The design of information rules requiring provision of information on produced environmental impacts or levels of environmental performance can also become an important motivator for the decision-makers to consider environmental impacts.

**Payoff rules**

The final group of the diagnostic framework is payoff rules. According to Ostrom and Crawford (2005:207) payoff rules ‘assign external rewards or sanctions to particular actions that have been taken or to particular readings on outcome state variables’. The IAD framework primarily links payoff rules to the net costs and benefits of actions for involved actors. However, as Ostrom and Crawford (2005:208) acknowledge these rules can incorporate a broader set of dimensions as ‘one could discuss payoff consequences of boundary rules tied to assignment of actions to positions, payoff consequences of information rules, as well as payoff consequences tied to choice and scope rules’. Consequently, in this group the study includes the rules which shape preferences of involved actors with regard to position uptake, choices of actions, consideration of cross-sectoral interests, application of environmental information and participation in joint decision-making. On the rule setting level, payoff rules shape preferences of involved actors to pursue particular solutions in the design of operational rules they or other actors are required or advised to observe.
The ecosystem management paradigm is built on the concept of ‘sustainability’ where social and economic goals are considered within the limits ‘dictated by the need to maintain a functioning life-support system’ (Fisher et al. 2007:622). However, in institutional arrangements pursuing ‘sustainable development’ goals, environmental considerations tend to be accommodated as competing and, more often than not, as conflicting interests. As McShane et al. (2011:970) argue, ‘win-win scenarios, where both natural resources are conserved and human well-being is improved in specific places over time, have been difficult, if not almost impossible, to realize.’ Furthermore, as the literature suggests, none of the governance actors, including local governments, private property owners and communities, can be assumed as having constant ‘intrinsic motivation’ to protect ecological systems and biodiversity (see section 4.4.1). Therefore, payoff rules shaping incentives for institutional actors to address environmental dimensions, or to resolve ‘trade-offs’, often become the main determinant of achieved environmental outcomes or impacts.

The overall payoff structure can be shaped by different configurations of rules. For example, on the appropriator/polluter/user level the choice of actions or compliance can be affected by perceived economic gains or losses linked to operating market structures, monetary incentives (e.g. subsidies, tradeable permits) or disincentives (e.g. taxes, levies, offsets), established sanctions (e.g. fines, loss of licence or permit), as well as costs related to participation in court proceedings. Preferences to enter these positions can also be affected by requirements to pay for licences or invest in specific technology (Ostrom 2005). On the governance level, motivation of the regulatory authorities to perform environmental duties can be affected by voting rules, budgeting rules, as well as rules providing for accountability or decision-making transparency (Doremus 2001, Lebel et al. 2006). Application, exchange or provision of environmental information can be shaped by data gathering costs, as well as funding distribution rules, which may alter incentives for agencies to cooperate or share data (Szaro et al. 1998a). Similarly, on the rule-setting level (transaction) costs related to the change of rules may affect whether operational rules will be changed in response to changing environmental conditions (Ostrom 2005).

The diagnostics of the payoff dimension can be a complex and challenging task. As outlined in chapter 2, determining sources of preferences of institutional actors and modelling their behaviour forms a significant part of the theoretical debate. Diversity of roles and interests, as well as the influence of different institutional and organisational rules may present significant challenges for locating and scoping relevant sets of rules affecting environmental preferences. Furthermore, preferences may not be constant and may change over time as a result of changed institutional roles, perception of the problem, interactions with other actors, or changes in the market structure (e.g. reduced/increased market demand for goods). Similarly, uncovering the payoff structure at the policy-making level can be a complex task, as numerous actors participate in the process and seek to influence outcomes based on their interests. In other
words, the presence of institutional misfits, which result from flawed behavioural models/assumptions, or ignorance of relevant sources of external (i.e. institutional) incentives or intrinsic values shaping human behaviour, could be as common as problems with modelling the behaviour of ecological systems.

In summary, institutional diagnostics combined with the typology of rules of the IAD framework enables the linking of diverse attributes of environmental problems into a unifying diagnostic framework. The framework is presented in Tables 5 and 6. Application of the diagnostic framework, its strengths and limitations is discussed in the final part of this chapter.
### Table 5 Diagnostic framework: operational rules

<table>
<thead>
<tr>
<th>Categories of rules</th>
<th>Environmental performance dimensions</th>
<th>Guiding questions</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Scope rules (SR)</strong></td>
<td>Specify:</td>
<td>To what extent:</td>
</tr>
<tr>
<td>Rules specifying what scope of outcomes and/or impacts resulting from what interventions, in what geographical area and within what timeframes institutional actor(s) are prescribed or permitted to achieve</td>
<td>- the desired state of the ecological system or its properties to be achieved or maintained as a result of managed interventions (e.g. water quality levels, maintained levels of biotic resources, retained habitat area)</td>
<td>▪ do the rules provide clear and consistent guidance:</td>
</tr>
<tr>
<td></td>
<td>- the levels of environmental performance the actors are required or permitted to achieve or observe (e.g. pollution levels, resource extraction limits)</td>
<td>- regarding regulated environmental outcomes or impacts, types of interactions, locations and timeframes;</td>
</tr>
<tr>
<td></td>
<td>- the scope of regulated interventions affecting ecological systems or their properties (e.g. vegetation clearing, conservation)</td>
<td>- how outcomes are prioritised (e.g. in case of conflicting goals)?</td>
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<tr>
<td></td>
<td>- spatial area where particular management interventions are prescribed, permitted or prohibited to occur or their impacts managed (e.g. local government area, management district, private property)</td>
<td>▪ are environmental outcomes prioritised over competing or conflicting outcomes?</td>
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<tr>
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<td>- timeframes for management interventions (e.g. seasons) or timeframes for the achievement of prescribed levels of environmental performance</td>
<td>▪ do the rules (e.g. rules specifying the desired state of the biophysical system, performance levels) consider:</td>
</tr>
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<td>- known characteristics of affected ecosystem or its properties (i.e. make ‘ecological sense’);</td>
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<td>- existing or potential pressures (including cumulative) resulting from the regulated scope and scale of management interventions;</td>
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<td>- existing or potential pressures produced by interventions regulated by other institutions (cross-boundary, cross-sectoral impacts) or lacking regulation?</td>
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<td>▪ do the established jurisdictional boundaries match (‘fit’) the spatial area required for the management of affected ecosystem or its properties?</td>
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<td></td>
<td>▪ does the scope incorporate interventions affecting achievement of desired state of the ecological system or its property (i.e. ‘leaves no gaps’)?</td>
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<tr>
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<td></td>
<td>▪ do established timeframes consider temporal scales relevant for the management of affected ecosystems or its properties (e.g. lag effects, life cycles of species)?</td>
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</table>
Position-boundary rules (PBR)

<table>
<thead>
<tr>
<th>Rules specifying:</th>
<th>Specify:</th>
<th>To what extent:</th>
</tr>
</thead>
<tbody>
<tr>
<td>- which and how many positions are established;</td>
<td>• which and how many positions are established with authority to:</td>
<td>• do the rules provide clear and consistent guidance regarding:</td>
</tr>
<tr>
<td>- when, how and under what conditions participants are prescribed or permitted to enter or leave these positions</td>
<td>- interact with ecosystems or their properties (e.g. position of resource users, polluters, appropriators);</td>
<td>- what and how many positions are established;</td>
</tr>
<tr>
<td></td>
<td>- coordinate, permit or restrict interactions with ecosystems, distribute resources and/or monitor performance (e.g. position of management/regulatory authority)</td>
<td>- how, for how long and under what conditions participants can enter or leave these positions?</td>
</tr>
<tr>
<td></td>
<td>- influence environmental decision-making process or outcomes (e.g. position of expert, advisor, stakeholder).</td>
<td>• do the rules require or enable the allocation of regulatory authority to actors who:</td>
</tr>
<tr>
<td></td>
<td>• eligibility requirements for participants to enter the position (e.g. knowledge level, capacity, representation, ownership rights)</td>
<td>- have the necessary capacity to undertake the allocated scope of actions;</td>
</tr>
<tr>
<td></td>
<td>• process by which participants become position holders (e.g. nomination, appointment, permit approval, registration of ownership rights)</td>
<td>- have a commitment or responsibility to protect environmental values;</td>
</tr>
<tr>
<td></td>
<td>• for how long participants are prescribed or permitted to hold the position and under what conditions participants are prescribed or permitted to leave (e.g. elected management bodies, temporal permit holders)</td>
<td>- hold relevant environmental information/knowledge or have the capacity to obtain required information/knowledge;</td>
</tr>
</tbody>
</table>

To what extent:

- do the rules require or enable the allocation of user/polluter/appropriator positions to participants motivated to sustain ecosystems or their properties?
- do the rules provide for positions authorised to influence environmental decision-making process or outcomes to stakeholders who:
  - benefit from the services of affected ecological systems or their properties, including adjacent and otherwise connected systems/properties;
  - hold responsibility for the management of other interactions with affected ecological systems/properties (e.g. other resource management authorities);
  - hold environmental information or knowledge?
<table>
<thead>
<tr>
<th>Choice Rules (CR)</th>
<th>Specify:</th>
<th>To what extent:</th>
</tr>
</thead>
</table>
| Rules specifying the scope of actions that institutional actors(s) in specified positions are prescribed, permitted or prohibited to undertake in light of particular conditions | - the scope of actions the actors in resource user/polluter/appropriator positions are permitted or prohibited to undertake  
- the scope of actions the actors in regulatory/management positions are required, permitted or prohibited to undertake  
- the state of environmental variables or other enabling or constraining conditions that the actors are prescribed or permitted to observe in their choice of actions (e.g. extraction limits under drought conditions, levels of environmental harm)  
- what actions under what conditions the actors holding stakeholder/advisor positions are prescribed or permitted or to undertake | - do the rules provide clear and consistent guidance regarding the allocated scope of actions and enabling and constraining conditions?  
- do the rules specifying the scope of actions for user/polluter/appropriator positions consider:  
  - known functional characteristics of affected ecological systems or their properties (e.g. habitat requirements, species lifecycle, productivity);  
  - existing or potential pressures (including cumulative) resulting from the regulated scope and scale of interventions;  
  - existing or potential pressures produced by interventions regulated by other institutions (cross-boundary, cross-sectoral impacts) or lacking regulation?  
- do the rules specifying the scope of regulatory/management functions (authority level) require or permit the regulatory body to:  
  - undertake actions directed towards protection or preservation of ecological systems/properties (e.g. establish protected areas, modify resource allocation limits, prohibit or constrain development);  
  - coordinate or monitor actions of institutional actors holding right to interact with or adversely affect ecosystems or their properties;  
  - impose and enforce sanctions (in case of violation)?  
- does the allocated scope of regulatory/management authority requires or enables to reduce pressures on ecological systems or their properties in response to:  
  - observed or predicted changes in the state of the biophysical environment or changes in the environmental knowledge (e.g. climate change);  
  - uncertainty in system’s response (e.g. precautionary approach);  
  - changes in pressures produced by interventions regulated by other institutions (cross-boundary, cross-sectoral impacts) or lacking regulation;  
  - changes in pressures produced to connected ecological systems/ properties?  
- does the allocated scope of actions enables stakeholders/advisors or wider public to constrain or dispute decisions of users or decision-making bodies to prevent or minimise adverse environmental impacts? |
<table>
<thead>
<tr>
<th><strong>Aggregation rules (AR)</strong></th>
<th>Specify:</th>
<th>To what extent:</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rules specifying which actors are involved in joint decision-making, how decisions are made and what level of control each actor has over the final decision</td>
<td>- which actors in positions are required or permitted to be involved in joint decision-making with regard to the achievement or prevention of particular environmental outcomes or impacts&lt;br&gt;- how joint decisions are made or agreement is reached (e.g. consensus, majority voting, veto rights)&lt;br&gt;- the level of control each actor in position has over the final decision</td>
<td>- do the rules provide clear and consistent guidance about:&lt;br&gt;  - which actors are involved in joint decision-making on the environmental outcomes or impacts;&lt;br&gt;  - how decisions are made;&lt;br&gt;  - what level of control each actor has over the final decision?&lt;br&gt;- do the rules regulating joint decision-making process require or enable participation of actors who:&lt;br&gt;  - hold the authority over the management of other properties of ecological systems or of shared or connected systems;&lt;br&gt;  - hold environmental information/knowledge or have the capacity to obtain required information/knowledge;&lt;br&gt;  - benefit from the services of affected ecological systems, including adjacent and otherwise connected systems or their properties?&lt;br&gt;- do the rules allocate the control over the final decision to actors who have a commitment or responsibility (e.g. institutional role) to protect environmental values?</td>
</tr>
</tbody>
</table>
### Information rules (IR)

<table>
<thead>
<tr>
<th>Rules specifying what information, in what form, in what timeframes, from what sources and through what channels the actors are prescribed or permitted to obtain, apply or provide</th>
<th>Specify:</th>
<th>To what extent:</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>- what environmental information the actors are prescribed or permitted to obtain, apply or provide to:</td>
<td>▪ do the rules provide clear and consistent guidance regarding what environmental information from what sources in what timeframes the actors are required or permitted to obtain, apply or provide?</td>
</tr>
<tr>
<td></td>
<td>▪ evaluate the progress towards the achievement of specified level of performance (e.g. state of environment, pollution levels)</td>
<td>▪ does the prescribed environmental information enable the actors to:</td>
</tr>
<tr>
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<td>▪ make decisions on actions;</td>
<td>▪ - understand the situation (e.g. assess impacts, predict the response to interventions etc.);</td>
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<tr>
<td></td>
<td>▪ inform other actors or the wider public on existing or potential environmental impacts of chosen actions or performance levels (e.g. environmental impact assessments)</td>
<td>▪ - evaluate the progress towards the achievement of specified level of performance (e.g. causal relevance of environmental indicators)?</td>
</tr>
<tr>
<td></td>
<td>▪ what environmental information, how often, in what form and through what channels the actors are prescribed or permitted to exchange</td>
<td>▪ do the rules require the actors in user/polluter/appropriator positions to obtain and provide information:</td>
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<tr>
<td></td>
<td></td>
<td>▪ - on the impacts of their actions on the managed ecological system or its properties;</td>
</tr>
<tr>
<td></td>
<td></td>
<td>▪ - on the impacts of their actions on connected ecological systems or their properties?</td>
</tr>
<tr>
<td></td>
<td></td>
<td>▪ do the rules require or permit the actors in regulatory positions to obtain, request or provide information:</td>
</tr>
<tr>
<td></td>
<td></td>
<td>▪ - on past, existing or potential environmental pressures (including cumulative) on the regulated/managed ecological system/property or connected ecological systems/properties (i.e. broader consequences of regulation);</td>
</tr>
<tr>
<td></td>
<td></td>
<td>▪ - on past, existing or potential pressures on the regulated/managed ecological system/property produced by interventions regulated by other institutions (cross-boundary, cross-sectoral impacts) or lacking regulation?</td>
</tr>
<tr>
<td></td>
<td></td>
<td>▪ do the rules specifying the scope of environmental information consider (i.e. ‘match’):</td>
</tr>
<tr>
<td></td>
<td></td>
<td>▪ - the capacities of users/regulatory authorities to obtain required information (e.g. financial resources, available expertise, developed data base) within established timeframes;</td>
</tr>
<tr>
<td></td>
<td></td>
<td>▪ - the capacities of the decision-makers to understand or process the required information (e.g. level of environmental expertise, education);</td>
</tr>
<tr>
<td></td>
<td></td>
<td>▪ - the capacities of knowledge holders (advisors) to provide required information (e.g. level of existing knowledge, involved uncertainties, level of expertise)?</td>
</tr>
</tbody>
</table>
## Payoff rules (PR)

<table>
<thead>
<tr>
<th>Rules specifying the incentives and disincentives the actors must or may encounter in:</th>
<th>Specify:</th>
<th>To what extent do the rules:</th>
</tr>
</thead>
<tbody>
<tr>
<td>- the choice of actions or performance levels; - position uptake; - application or provision of information; or - participation in joint decision-making process</td>
<td>▪ incentives or disincentives for actors in relation to: - the choices of actions affecting ecological systems or their properties; - the choices of performance levels (e.g. levels of pollutant releases, extraction levels, allocated areas for development) ▪ the incentives or disincentives for actors (participants) in relation to the uptake of positions authorised to: - affect (use or destroy) ecosystems or their properties, - regulate or monitor environmental impacts; - influence environmental decisions (e.g. challenge user or management decisions). ▪ the incentives or disincentives for actors with regard to the application, generation, provision or exchange of environmental information ▪ the incentives or disincentives for actors to engage in joint decision-making regarding the environmental outcomes or impacts</td>
<td>▪ provide clear and consistent guidance regarding incentives or disincentives for actors regarding: - the choices of actions affecting ecological systems or their properties; - the choices of environmental performance levels; - uptake of regulatory, user/polluter/appropriator or stakeholder positions; - application or provision of environmental information; - participation in joint decision-making process? ▪ reduce incentives for participants to uptake or hold positions authorised to adversely affect ecological systems or their properties (e.g. payment for licences, investment)? ▪ provide incentives for actors in user/polluter/appropriator positions to: - avoid or minimise adverse impacts on ecosystems or their properties (e.g. biodiversity offsets, subsidies, payments for ecosystem services); - comply with established restrictions or undertake the scope of prescribed actions; - obtain or provide information on existing or potential environmental impacts of chosen actions or performance levels to regulatory authority or wider public? ▪ provide incentives for participants to uptake positions influencing environmental decision-making process (e.g. provide expertise, challenge decisions)? ▪ provide incentives for institutional actors in regulatory-management positions to: - prioritise environmental values or minimise trade-offs leading to permanent loss or destruction of environmental values, including biodiversity; - prohibit or constrain the scope of users’ actions adversely affecting ecological systems/properties or monitor or enforce compliance; - apply, generate or update environmental information required to support decision-making; - provide information on environmental effects of their actions or decisions to other institutional actors (e.g. other management bodies) or wider public; - consider effects on environmental values held by other stakeholders; - engage in cross-boundary or cross-sectoral cooperation or collaboration?</td>
</tr>
</tbody>
</table>

Source: author
<table>
<thead>
<tr>
<th>Types of rules</th>
<th>Environmental performance dimensions</th>
<th>Guiding questions</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Scope rules: Collective choice level (SRCC)</strong></td>
<td>Specify:</td>
<td></td>
</tr>
</tbody>
</table>
| Rules specifying what interventions for what geographical area, in what timeframes the actors are prescribed or permitted to regulate | - the scope of interventions with ecological systems or their properties involved actors are prescribed or permitted to regulate  
- geographical area for which interventions can be regulated (e.g. local government area, resource management district, coastal zone)  
- the scope of actors whose actions can be regulated (e.g. freehold or leasehold land owners, resource appropriators)  
- when or in what timeframes the actors are required or permitted to develop or change regulation | To what extent:  |
| | | - do the rules provide clear and consistent guidance regarding the scope of regulation?  
- does the allocated regulatory scope require or enable involved actors to:  
  - consider biodiversity protection outcomes;  
  - prioritise biodiversity protection over competing or conflicting outcomes;  
  - cover the scope of interventions that produce adverse effects within the issue-area domain of regulation (i.e. leaves no gaps);  
  - regulate impacts on connected ecological systems or their properties?  
| | | - do the established jurisdictional (geographical) boundaries match the spatial area required for the regulation of specified interactions?  
| | | - do the allocated regulatory scope and timeframes enable the actors to respond to:  
  - observed or predicted changes in the biophysical environment or changes in the environmental knowledge or values;  
  - changes in actions of other actors producing pressures on shared ecosystems? |
| **Position-boundary rules: Collective choice level (PBRCC)** | Specify:  |
| Rules specifying positions with regulatory authority and conditions under which participants can enter or leave these positions | - what and how many positions are established with the authority to perform functions of environmental regulator  
- what and how many positions are established with the authority to influence regulatory process (e.g. advisor, expert, stakeholder)  
- the eligibility requirements for participants to enter these position (e.g. knowledge level, capacity, representation, legal standing)  
- how participants become position holders and under what conditions they can leave the position (e.g. elected representatives) | To what extent do the rules:  |
| | | - provide clear and consistent guidance regarding the distribution of regulatory positions, eligibility requirements, position allocation and change process?  
| | | - enable or require position uptake (i.e. establish eligibility requirements) by participants who:  
  - have a commitment or responsibility (e.g. institutional role) to protect environmental values;  
  - hold relevant environmental information/knowledge or have the capacity to obtain, required information/knowledge;  
  - represent the community having interest in protection of environmental values or benefiting from the services (e.g. consumptive, non-consumptive use) of affected ecological systems, including adjacent and otherwise connected systems? |
### Choice rules: Collective choice level (CRCC)

<table>
<thead>
<tr>
<th>Rules specifying what scope of regulatory instruments under what conditions the actors are prescribed or permitted to develop or change</th>
<th>Specify:</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>▪ the scope of regulatory instruments the actors in regulatory positions are required or permitted to develop or change to achieve environmental outcomes</td>
</tr>
<tr>
<td></td>
<td>▪ what scope of actions under what conditions the actors in stakeholder positions are required or permitted to undertake to influence the development or change of regulatory instruments</td>
</tr>
</tbody>
</table>

| | To what extent do the rules: |
| | ▪ provide clear and consistent guidance regarding what scope of regulatory instruments under what conditions responsible actors are required or permitted to develop or change? |
| | ▪ require or enable institutional actors in regulatory positions to develop or change mechanisms which: |
| | - confer protective status on ecosystems or their properties; |
| | - limit or prohibit actions which might adversely affect ecosystems or their properties; |
| | - enable consideration of cross-boundary effects (e.g. adjacent ecosystems) or effects on other properties of the system; |
| | - enable consideration of pressures on the ecological system or its properties produced by other interventions regulated or lacking regulation; |
| | - provide for incentives for users/polluters/appropriators to protect ecosystems and biodiversity? |
| | ▪ require or enable the actors in regulatory positions to engage in collaboration, cooperation or coordination with other institutional actors involved in the regulation of shared ecological systems or properties? |
| | ▪ enable the actors in stakeholder positions to influence rule setting process towards protection of environmental values? |
### Aggregation rules: Collective choice level (ARCC)

Rules specifying which institutional actors are involved in joint decision-making regarding the development or change of regulatory instruments, how decisions are made and what level of control each actor has over the final decision

<table>
<thead>
<tr>
<th>Specify:</th>
<th>To what extent do the rules:</th>
</tr>
</thead>
<tbody>
<tr>
<td>- which institutional actors must or may be involved in the joint (collective) decision-making regarding the development or change of regulatory instruments providing for environmental outcomes</td>
<td>- provide clear and consistent guidance regarding:</td>
</tr>
<tr>
<td>- the level of authority or control each actor has over the final decision</td>
<td>- which actors are authorised to participate in joint decision-making regarding the development or change of regulatory instruments;</td>
</tr>
<tr>
<td>- how decisions are made or the agreement is reached (e.g. consensus, majority voting)</td>
<td>- what level of control each actor has over the final decision;</td>
</tr>
<tr>
<td></td>
<td>- how the agreement is reached?</td>
</tr>
</tbody>
</table>

### Information rules: Collective choice level (IRCC)

Rules specifying what environmental information the actors are prescribed or permitted to apply, generate or provide in the process of the development or change of regulatory instruments

<table>
<thead>
<tr>
<th>Specify:</th>
<th>To what extent:</th>
</tr>
</thead>
<tbody>
<tr>
<td>- what environmental information the actors are prescribed or permitted to obtain, apply or provide to evaluate the performance of established regulatory instruments (e.g. environmental performance criteria, indicators)</td>
<td>- do the rules provide clear and consistent guidance regarding what environmental information needs to be obtained or provided in the assessment, development or change of regulatory instruments providing for environmental outcomes?</td>
</tr>
<tr>
<td>- what environmental information the actors are prescribed or permitted to apply or exchange to support the development or change of regulatory instruments providing for environmental outcomes</td>
<td>- do the rules require or enable the actors to obtain or provide the information:</td>
</tr>
<tr>
<td></td>
<td>- on past, existing or potential impacts (pressures) of regulated interventions on ecological systems or their properties;</td>
</tr>
<tr>
<td></td>
<td>- on past, existing or potential impacts (pressures) produced by other interventions (i.e. outside the regulatory scope) on affected ecological systems or their properties?</td>
</tr>
<tr>
<td></td>
<td>- does the specified environmental information enable institutional actors to capture decision-making situation and evaluate potential environmental outcomes and/or impacts produced by particular regulatory solutions?</td>
</tr>
</tbody>
</table>
### Information rules: Collective choice level (IRCC) continued

<table>
<thead>
<tr>
<th>Specify:</th>
<th>To what extent:</th>
</tr>
</thead>
<tbody>
<tr>
<td>- environmental information (type, form, applicable sources) that the actors are required or permitted to provide to other regulators or the wider public (e.g. strategic impact assessments) on potential environmental impacts of regulation</td>
<td></td>
</tr>
<tr>
<td>- how environmental information is clarified or generated and how information conflicts are resolved (e.g. under conditions of uncertainty or limited knowledge)</td>
<td></td>
</tr>
<tr>
<td>- do the rules providing for information/knowledge generation process enable involved actors to:</td>
<td></td>
</tr>
<tr>
<td>- reduce uncertainty and gain shared understanding about the environmental problem under consideration;</td>
<td></td>
</tr>
<tr>
<td>- resolve information conflicts or avoid manipulation with environmental information?</td>
<td></td>
</tr>
</tbody>
</table>

### Payoff rules: Collective choice level (PRCC)

<table>
<thead>
<tr>
<th>Rules specifying the incentives and disincentives for the actors undertaking the assessment, development or change of regulatory instruments</th>
<th>Specify the incentives and disincentives:</th>
</tr>
</thead>
<tbody>
<tr>
<td>- the actors (participants) must or may face in relation to the uptake of positions authorised to:</td>
<td></td>
</tr>
<tr>
<td>- assess, develop or change regulatory instruments providing for environmental outcomes;</td>
<td></td>
</tr>
<tr>
<td>- participate in or influence (e.g. as stakeholders or experts) the process of assessment, development or change of regulatory instruments</td>
<td></td>
</tr>
<tr>
<td>- the actors must or may face in relation to the choice of particular regulatory instruments providing for environmental outcomes (e.g. instruments providing for protection of environmental values)</td>
<td></td>
</tr>
<tr>
<td>- the actors must or may face with regard to the application, provision or exchange of environmental information</td>
<td></td>
</tr>
<tr>
<td>- provide clear and consistent guidance regarding what incentives/disincentives the actors must or may face in the assessment, choice or change of specified regulatory instruments providing for environmental outcomes?</td>
<td></td>
</tr>
<tr>
<td>- provide incentives for institutional actors holding (individually or jointly) regulatory authority to:</td>
<td></td>
</tr>
<tr>
<td>- consider environmental values and adopt solutions which minimise trade-offs leading to permanent loss or destruction of ecological systems and biodiversity;</td>
<td></td>
</tr>
<tr>
<td>- engage in cross-boundary or cross-sectoral coordination or consultation;</td>
<td></td>
</tr>
<tr>
<td>- consider environmental values held by other stakeholders and wider public;</td>
<td></td>
</tr>
<tr>
<td>- consider environmental information on the state of affected biophysical systems and potential environmental outcomes or impacts;</td>
<td></td>
</tr>
<tr>
<td>- consider uncertainty and limits to knowledge (e.g. apply precautionary approach);</td>
<td></td>
</tr>
<tr>
<td>- respond to changing environmental conditions or accumulated environmental knowledge?</td>
<td></td>
</tr>
<tr>
<td>- provide incentives for stakeholders having interest to protect environmental values and/or holding relevant knowledge to participate (i.e. uptake positions) in the assessment, development or change of specified regulatory instruments?</td>
<td></td>
</tr>
</tbody>
</table>

Source: author
4.3.2 Diagnostic framework: application, limitations and potential

This chapter has proposed a sound framework for the systematic evaluation of institutions structuring human interactions with ecological systems. Its primary value lies in the application of an existing, empirically tested classificatory system of rules to organise the analysis of diverse determinants of institutional environmental performance. Consistent with the aim of this study, the framework has been designed with a focus on formal institutional systems (i.e. written rules) operating at the state level and providing for ongoing regulation of human–environment interactions. At the same time, it is recognised that the framework cannot cover the full diversity of institutional settings, and may require adjustment or expansion of categories of rules and diagnostic questions. This section discusses the application, limitations and potential of the framework.

Application

The framework is designed to incorporate rules structuring two hierarchical levels of interactions, which, using the vocabulary of the IAD framework, are described as ‘operational’ and ‘collective-choice’ rules. However, the groups of rules (performance dimensions) are not linked to specific environmental decision-making situations, groups of actors or governance levels. This study recognises that institutions, particularly those operating at the state level, are highly connected and isolating any one specific ‘arena’ as fully accounting for environmental outcomes is almost impossible. As McGinnis (2011) points out several operational action situations may interact simultaneously and any of them may be affected by multiple processes at a collective, as well as constitutional-choice level.

The rules operating as determinants of environmental outcomes can be the output of numerous policy-making situations involving the same, partially overlapping or absolutely different groups of actors. Many aspects of the environmental problem, such as cross-boundary linkages, cumulative effects, or capacity or motivation of the actors, may require the examination of rules incorporated in different statutory or non-statutory frameworks. In practice, so called ‘institutional connectivity’ (Andersson & Ostrom 2008), ‘interplay’ (Young 2002), or ‘networks of adjacent action situations’ (McGinnis 2011) complicate the scoping of rules for analysis. It challenges conventional approaches to drawing assessment boundaries, based on a single statute, related groups of regulatory instruments, or rules established by particular groups of actors (e.g. community rules, local government laws). It can be argued that in the case of ‘networks’ the effectiveness of any statute, policy or program can be described as the extent to which established configurations of rules complement (i.e. ‘fit in’) or modify the relevant network to produce particular environmental outcomes. As a result, the diagnostic framework is intended to be flexible and adjustable to suit various assessment purposes. It can be employed to
explore interactions between several groups of rules operating at the same level, or across
several levels of the hierarchy.

‘Institutional connectivity’ has also affected the design of the diagnostic framework. In practice,
there is a limited number of cases where the whole scope of operational rules guiding ‘on
ground’ interactions with the biophysical systems can be the output of one or few policy-
making arenas. However, initial attempts made in this study to consider each category of
operational rules as an output of a separate action situation, which is structured by a separate set
of collective-choice rules, produced a highly complicated structure. Therefore for simplicity, the
framework has been designed assuming that all operational rules are an output of a single
policy-making situation, affected by a single set of collective-choice rules. Consequently,
adjustments in the description of relevant performance dimensions may be required depending
on selected policy-making arenas. Which collective-choice situations and respective sets of
rules need to be considered cannot be prescribed without considering the particulars of each
case.

The main purpose of the diagnostic questions is to assist with the selection and analysis of
relevant groups of rules and to identify factors which may affect the potential of particular
institutional designs to contribute to environmental outcomes. The questions have been
designed to address two aspects of institutional performance. The first is the ‘quality’ of rules.
Clarity and consistency of guidance is an important determinant of behavioural change in any
institutional setting. As Ostrom (2005:20) points out regularities in actions ‘cannot result if
those who must repeatedly interpret the meaning of a rule within action situation arrive at
multiple interpretations.’ The second covers specific aspects of institutional performance which,
according to the theoretical and empirical literature, have been identified as most relevant for
the management of complex environmental problems, including those conceptualised as
problems of ‘institution – ecosystem fit’. Consistent with the focus on institutional
effectiveness, diagnostic questions have been formulated in terms of the ‘extent’ to which
particular groups of rules incorporate or consider specific problem attributes, prescribe or enable
interactions, or contribute to the achievement of particular outcomes, outputs or impacts.

Limitations

Taking into account the diversity of problem attributes and institutional designs the framework
is presented with several limitations. First, this study makes no claims that the proposed
dimensions are the exclusive determinants of institutional ‘success’ or ‘failure’ in the
management of human-environment interactions. The role that particular rules play in the
management of a specific problem needs to be identified on a case-by-case basis. Second, to
maintain some level of general applicability, the framework contains only a broad description of
rules. Therefore, each group of rules can be divided further into sub-categories, or modified to
suit a particular management problem, set of involved actors or type of interventions (e.g. pollution, resource extraction).

As Ostrom and Crawford (2005:191) point out, rules operate together ‘as a configuration’. Despite the attempt to address the linkages between separate groups of rules (see e.g. payoff rules) or alternative choices to regulation, the framework cannot cover the whole scope of potential interactions. For example, a high level of discretion (i.e. decision-making flexibility) tends to be identified in the literature as a ‘positive’ feature to address environmental uncertainty. However, when coupled with a flawed accountability system it may potentially produce more environmental problems than a restricted and rigid scope of authority (Doremus 2001, Lebel et al. 2006). Similarly, improvements in payoff rules, achieved through such instruments as biodiversity offsets, may produce adverse effects if institutional design fails to consider the nature of affected ecological properties. Therefore, the quality of the diagnostic exercise will depend on the ability of the analyst to identify relevant linkages between separate categories of rules and to explore how they may affect interactions and outcomes.

Similarly, the set of diagnostic questions needs to be approached critically for several reasons. First, the framework cannot be applied as an assessment ‘checklist’ to determine the extent to which a particular institution ‘matches’ listed aspects. Not all questions can be answered and not all performance aspects will be relevant for specific environmental problems. For example, alignment of jurisdictional and ecological boundaries can be relevant in resource (over-) extraction cases (e.g. water management), but is almost impossible if the regulatory scope involves protection of species having diverse or large habitat ranges. Second, some of the identified performance aspects need to be approached with caution. Numerous studies tend to refer to institutional barriers, or include proposals for institutional change, without relying on a sound empirical basis or consideration of other institutional and non-institutional variables that may contribute to a particular problem. While this study takes a critical approach to examining different propositions, it cannot test under what conditions they will ‘work in practice’. Further research is required to create a sound knowledge base which could support the diagnostics exercise.

The framework does not provide any guidance regarding methodology that can be applied to evaluate institutional effectiveness along particular dimensions or their configurations. Diversity of problem attributes, and the ways particular groups of rules can affect separate elements of the decision-making situation, suggest a large variety of applicable criteria as well as approaches to data gathering and analysis. While the questions have been formulated in terms of the ‘extent’ to which particular aspects of environmental performance have been considered or achieved they do not imply particular measurement scales. In this context, perhaps the biggest challenge is the need to work across concepts, theories and models from different disciplines of both social and environmental sciences.
Finally, the application of the framework does not replace the need to explore the structure of the environmental problem underpinning institutional design on a case-by-case basis. The framework cannot define all problem attributes and their characteristics, assessment boundaries (i.e. required ‘network of rules’) or critical sets of diagnostic questions. What types of actors need to be considered, how to characterise their behaviour, what models or theories, if any, can be applied to explain or predict behavioural change, can be either relatively simple or a highly complex task. Similarly, environmental problems can range from ones which are obvious and easy to monitor (e.g. clear-cutting of forests, land transformation) to those which require understanding of complex biophysical and economic models underpinning institutional design (e.g. fisheries or water extraction systems). In some cases, assessment can be confined to a relatively isolated set of rules, whereas in others ignorance of complex rules networks affecting focal decision-making situation can lead to serious mistakes in ‘diagnosing’ the source of the problem. As Ingram et al. (1984:333) summarised:

Institutional analysis, because it deals with complexities and dynamics, is time consuming, intellectually challenging and costly. It cannot be done ‘on the cheap’, it cannot be done with inadequate tools; and it cannot be purchased in a ‘canned’ form from work done elsewhere.

**Potential**

Despite the limitations, this study argues that the proposed framework is helpful for both theoretical and empirical research focused on detailed analysis of complex institutional structures providing for ongoing human-environment interactions. It is one of the few promising attempts to build a bridge between the growing body of the literature analysing different aspects of environmental problems and institutional studies. To this end, the framework provides the means to unpack institutional complexity to deal effectively with the diverse determinants of environmental outcomes. It draws on the argument frequently emphasised by current institutional analysts (Dovers 2001b, Ostrom 2005, 2007, Young 2011) that the complexity of institutional designs cannot be reduced to a few groups of rules. As Dovers (2001a:8) points out, ‘ignoring complexity lessens the chance of matching specific institutional capacities with specific problems and contexts’.

The framework has demonstrated that a large set of institutional variables addressed in the environmental science, ecosystem management and environmental governance literature can be organised using the generic typology of rules of the IAD framework. It also offers a good basis for comparative studies and accumulation of empirical knowledge. The framework is ‘ecocentric’ and predominately addresses environmental aspects of institutional performance. To produce more comprehensive assessments it can be further expanded to incorporate other dimensions determining economic and social outcomes. While these aspects of institutional performance are beyond the scope of this study, the proposed process of the framework design can serve as a starting point for the development of more comprehensive assessment designs.
4.4 Summary and conclusions

This chapter completes the first part of this study addressing the effectiveness assessment of institutional environmental performance. The core question examined in this chapter was how to examine a large variety of institutional determinants of the biodiversity protection problem in a logical and structured way. More specifically, it addressed two major challenges for assessment design: the variety of dimensions that can operate as determinants of institutional environmental performance in different settings, and the diversity of institutional designs.

There are no universal sets of rules or performance dimensions that can account for successful environmental performance across the vast number of problems and contexts. To identify which design features characterise institutional potential to produce or resolve environmental problems in specific settings, this chapter examined an application of the institutional diagnostics. As described by Young (2002, 2008) institutional diagnostics is a systematic process for examining different problem attributes or diagnostic conditions to specify institutional arrangements that could address them. It was concluded that, despite problems with the knowledge base, an application of institutional diagnostics offers the best means for examining a large diversity of problem attributes in a structured way.

The diversity of institutional designs presents the problem of how to disaggregate these complex structures into common groups of features. This chapter examined the structure of the Institutional Analysis and Development (IAD) framework and its typology of rules as proposed by Ostrom and Crawford (2005). Drawing on the game-theoretic approach, the IAD framework satisfactorily disaggregates the complex institutional hierarchy into the sets of rules common across different decision-making situations. This chapter demonstrated that, used in combination with the diagnostics process, the IAD framework offers a sound foundation for linking and structuring diverse institutional determinants of environmental problems into a diagnostic framework.

Thus the main output of the first part of this study is a diagnostic framework for the analysis of different institutional systems regulating human-environment interactions. It has been designed to examine several groups of rules interacting at two vertical levels of hierarchy, and cumulatively affecting ‘on ground’ behavioural outcomes and resulting environmental impacts. Building on a generic typology of the ‘rules of the game’, the framework has been tailored for the analysis of formal institutions operating at the state level. It has been designed to assist the analysts to identify and map relevant configurations of rules, and to examine how they enable or constrain the resolution or creation of environmental problems in specific settings. This framework is summarised in Tables 5 and 6.

The next part of the study applies the framework in an evaluation of the environmental performance of the Queensland Planning System.
Chapter 5 - Queensland Planning System: Environmental Performance Evaluation, Part I

5.1 Introduction

This chapter begins the second part of the thesis. It is guided by an overarching question regarding the effectiveness of the Queensland land use planning and development assessment system (Queensland Planning System) in achieving biodiversity protection outcomes.

Australia has a complex governance system with three tiers of government having different regulatory powers and various administrative bodies, both governmental and private, performing different environmental planning and management functions. Their interactions are structured by a large number of agreements, laws, regulations, policies and strategies. Most environmental matters addressed in the overall regulatory system are either directly or indirectly linked to land use planning and development assessment activities and need to be considered in the analysis. Taking into account the broad scope of data, the analysis of the Queensland Planning System is presented in two parts in chapter 5 and chapter 6. This chapter examines the broader institutional setting which determines the scope and nature of regulatory solutions in environmental matters in Australia and in the state of Queensland.

The chapter consists of four major parts. It starts (section 5.2) with a description of the methodological approach applied in the analysis of the Queensland Planning System. The second part (section 5.3) describes the problem of biodiversity loss in Australia and in the state of Queensland and outlines major drivers of the use of land and extractive environmental resources and the resulting pressures on the natural environment. The third part (sections 5.4 - 5.6) describes the institutional system. It starts with an introduction to institutional foundations determining power relations between the national, state and local governments. Next, it describes the overall structure of the environmental governance and its institutional determinants. It examines the distribution of regulatory roles and responsibilities among the federal, state and local governments and details organisational and institutional arrangements in the state of Queensland. It concludes with a description of the land tenure and resource allocation system and available regulatory instruments to promote biodiversity protection. Finally, the fourth part (section 5.7) discusses major strength and weaknesses of the overall regulatory framework affecting the achievement of biodiversity protection outcomes in the state of Queensland.
5.2 Methodological approach

The overall purpose of the evaluation study is to establish whether the Queensland Planning System has the potential to prevent the loss of biodiversity resulting from the use of land resources at the state scale. Based on the diagnostic framework (Tables 5 and 6), this and the following chapter examine how different dimensions of institutional performance contribute to the creation or resolution of the problem.

The design of the evaluation study was not a linear process. The methodological approach evolved over time as a result of an ongoing iteration between identified elements of the assessment design (see Figure 3 in chapter 2), and attempts to create a sound structure capturing the complexity of a multi-scalar institutional system. This has resulted in both a gradual refinement of the approach and an understanding of gaps and limitations of institutional analysis in ‘real life’ settings. The discussion of methodological challenges encountered in the analysis of complex institutional systems and the applicability of the IAD-based diagnostics is included in chapter 7.

This section presents the final approach taken to the design of the evaluative study. It describes the overall structure of the study, analysis timeframes, data sources, analytical approach and limitations. Additional details specifically relating to the structure and analysis of the environmental performance of the Sustainable Planning Act 2009 (Qld) (SPA) are provided in chapter 6 (section 6.2).

5.2.1 Structure of the study

As Creswell (2013:249) notes, ‘the proper case to be studied is both ‘bounded’ and a ‘system’.’ In institutional assessment, delineation of institutional boundaries (i.e. specification of the object of the analysis) establishes the basis for evaluation. Moreover, as outlined in chapter 2, the scope of rules selected for evaluation may significantly affect analytical outcomes.

Diverse horizontal and vertical linkages between different regulatory sub-systems present significant challenges for analysing Australian environmental statutes as ‘stand-alone’ systems. In practice, environmental regulation cannot be approached as a separate system. In many aspects its structure and regulatory approaches depend on the broader institutional setting that establishes foundations for power and resource distribution. In this context, a key challenge is to create a sufficient, as well as a ‘manageable’ (in data terms) knowledge base of relevant institutional arrangements.

The Queensland Planning System is a complex and dynamic institutional system. As a result, the evaluation has been approached from two perspectives, which has divided the study into two major parts:
From the broader perspective, the Queensland Planning System is viewed as a complex interacting system of rules established at different governance levels, which cumulatively determine the current land use pattern in the state. This system sets out the overall context for the design and operation of any sub-system of rules regulating specific scope of environmental matters. This system is examined in chapter 5.

From the narrower perspective, the SPA is examined as a separate sub-system of rules regulating the particular scope of land use planning and development assessment decisions. The environmental performance of the SPA is linked to the provisions of the Vegetation Management Act 1999 (Qld) (VMA), which determines the extent of native vegetation cover at the state scale. The SPA-VMA framework is examined in chapter 6.

Both parts of the study are linked. Chapter 5 covers a broad scope of institutional arrangements influencing biodiversity protection outcomes in the state of Queensland. It examines the rules determining roles, authorities, capabilities and motivation of major governance actors and institutional ‘outputs’ produced to regulate various development pressures. Chapter 6 examines the SPA-VMA framework as a separate system ‘embedded’ in the overall regulatory framework. It builds on findings from the analysis of the broader regulatory framework.

The overall process employed to specify the object of the analysis can be described as ‘institutional mapping’ (Selin and VanDeveer 2003, Aligica 2006). It is an iterative and interactive process aiming to identify the relevant institutions’ roles and inter-institutional linkages (Aligica 2006). The scope of rules selected for the analysis has been determined by two factors. The first is the breadth of the problem domain, which determines the boundaries of the ‘arena’. The second concerns the causality linkages which specify what sub-sets of rules influence the interactions in the ‘arena’ and account for the resolution or creation of the problem of biodiversity loss.

To address the complexity of the problem domain, chapter 5 treats the concept of ‘arena’ very broadly. The overall boundaries of the study have been defined by the major drivers of the problem of biodiversity loss (see section 5.3), which determine the required scope of regulatory responses. As a result, the ‘biodiversity protection arena’ incorporates a broad range of actors and interactions related to both creation and mitigation of the problem at the state scale. To narrow down the scope of the analysis, this study predominately focuses on the rules impacting upon the behaviour of two core groups of actors, which are the regulators and resource owners.

The process of data (rules) scoping has been guided by six broad groups of questions listed in Table 7. They have been developed based on the diagnostic framework (Table 6) and the literature review. The aim of the questions is to assist with structured search for diverse institutional determinants of environmental problems as identified in the framework.
Table 7 Guiding questions for data scoping, Part I

(1) What are the core drivers of biodiversity loss in Australia and in the state of Queensland? Which categories of human activities or economy sectors impact upon biodiversity? What are their effects? What are the dominant land uses and how they contribute to biodiversity protection/loss?

(2) What is the distribution of regulatory power and revenue raising capacities among the governance actors? What are their power relations? What incentives/disincentives established system provides for the regulators to pursue biodiversity protection outcomes?

(3) What system of rights and responsibilities has been established with regard to the use of land and environmental resources? What incentives the established system provides for resource users to protect natural systems and biodiversity?

(4) What information systems have been established to gain understanding of the state of biodiversity? How the state is monitored and reported? What scientific platform (standards, criteria, best practices) underpins planning and implementation of biodiversity protection measures?

(5) Which tier of government has jurisdiction over the distribution and use of environmental and land resources? Which regulator plays the lead role in biodiversity protection? What measures have been adopted to coordinate different regulatory authorities and resolve conflicting matters?

(6) What regulatory measures have been adopted to prevent or mitigate actions adversely impacting upon the natural systems and biodiversity? How biodiversity protection is prioritised against competing land use interests? What institutional frameworks are in place to assign value to biodiversity and provided ecosystem services?

Source: author

The process of determining which institutional arrangements influence environmental outcomes involves identifying causal relations. Some of the relations examined in this analysis, in particular in Part I, have been established based on current theories guiding governance analysis (e.g. the role of private property regimes, market incentives, accountability systems). However, the current theoretical basis is far from sufficient to explain behavioural patterns emerging from multiple interacting sub-systems of rules. As a result, the scoping of rules was an interactive process. To identify the relevance of particular sub-systems of rules, and the nature of their effect, they were examined in the overall governance context. Causal relevance and the linkages were established, configured and re-configured during the data gathering and analysis process.

Figure 9 illustrates the approach. At its centre are the core groups of rules of the IAD framework (Ostrom 2005). Arrows from different sub-sets of rules examined in chapter 5 identify the nature of influence on decision-making situation.\(^\text{14}\)

\(^{14}\) Note: this figure is developed to illustrate the ‘logic’ applied in this study. It is not a comprehensive map of all institutional arrangements and does not account for all potential linkages.
Figure 9 Approach to scoping and structuring the rules determining the land use pattern and biodiversity outcomes in Queensland
Source: interior figure reprinted from Ostrom (2005:189), groups of rules added by author

Chapter 6 (Part II) draws the boundaries of the study based on the scope of the land use planning and development assessment activities regulated under the SPA. The SPA is analysed as a separate sub-system established to pursue certain goals and objectives within specified functional and geographical boundaries. The process of selecting relevant groups of rules for the analysis follows a similar logic model as described above. Detailed description of the approach taken to the structuring of the second part of the study is provided in chapter 6 (section 6.2).

5.2.2 Analysis timeframes

The analysis of any institutional system requires setting temporal boundaries. As outlined in chapter 2, the core point of reference is the configurations of rules as they exist at a selected point in time. A ‘snapshot in time’ view implies that an institutional system is analysed as an assumed ‘end-product’ of the institutional design process. Consequently, the effectiveness analysis, whether compared against historical points of reference or future projections (see Figure 2 in chapter 2), is conducted under the assumption that established rules remain unchanged.

Selection of fixed points of reference for the analysis of the Queensland Planning System encountered several challenges. Each regulatory sub-system examined in this and the following chapter had a different historical trajectory and major reform points. Furthermore, many of them were highly dynamic. They have been subject to frequent amendments driven by various factors, ranging from the need to respond to changing decision-making environment to the
requirement to implement election promises of the leading political party. For example, since its enactment in 2009 the SPA has been amended 28 times (as at March 2014). Furthermore, in 2014, the Queensland Government announced another major reform which is expected to result in the enactment of the Planning for Prosperity Act (DSDIP 2014a).

Institutional dynamics could not be ignored. As a result, the descriptive part in chapters 5 and 6 has been corrected and updated several times to reflect, as much as possible, the latest regulatory system in force at the final period of writing which was May 2014. In particular, the study had to consider recent changes in the political environment in the state of Queensland. In the 2012 elections the Australian Labor Party, which had been leading the Queensland Government for more than two decades, was defeated by the Liberal National Party (LNP). This change in political direction has led to extensive reforms of administrative and regulatory systems significantly affecting environmental protection measures. Consequently, many of the initial findings and conclusions required re-examination in the light of recent changes.

To address the problem of institutional dynamics, this study has adopted a flexible approach to temporal points of reference. Some institutional sub-systems have been examined in the light of their historical evolution. Others have been discussed in future terms based on the assumption of further implementation of proposed regulatory reforms. It is recognised that such an approach may not meet some of the criteria for clarity and consistency of the research design. However, situating the analysis within different temporal scales assists to gain a better understanding of the various trajectories of institutional development and change. It also helps to predict the longevity and direction of regulatory reforms.

5.2.3 Data sources, presentation and analysis

Data sources and presentation

The analysis of the Queensland Planning System is a desktop study. Data has been gathered by means of a document analysis which is a method commonly employed in case studies (Bowen 2009, Yin 2009). The information has been collected for two major purposes. The first was to gain an understanding of the links between particular regulatory sub-systems and their causal influence on biodiversity protection outcomes. The second was to distil a set of criteria and data characterising the nature and, if possible, the extent of produced or potential effects.

The object of the analysis is a formal institutional system. Therefore, the core sources of data are the Acts of Parliament, subordinate legislation, policy and strategy documents and other regulatory instruments. Statutes and subordinate legislation were retrieved from governmental websites. Commonwealth legislation was retrieved from the Australian Government database ComLaw (http://www.comlaw.gov.au/). The legislation of the state of Queensland was retrieved

15 Note: the use of other timeframes is indicated in the study.
from the database maintained by the Office of the Queensland Parliamentary Counsel (https://www.legislation.qld.gov.au/OQPChome.htm). Strategies, policies, planning instruments (e.g. regional plans and state planning policies), regulatory maps, assessment codes and other instruments were retrieved from the websites of administering governmental agencies and local governments.

The evaluative study draws on a broad range of documentary sources. Many findings are based on the government reports prepared or commissioned by the Commonwealth and Queensland Governments (e.g. state of environment reports, annual reports of governmental agencies, reports of governmental commissions etc.), and reports and data provided by the Australian Bureau of Statistics (ABS). Data was also obtained from governmental websites, commissioned research studies, summaries of public submissions, press releases, published opinion statements of non-governmental organisations (e.g. Environmental Defenders Office), and academic publications.

The use of a broad range of documentary sources offers two major advantages. First, it is the broad basis of data which allows cross-checking of findings and, thus, increases the validity of analytical outcomes (Bamberger et al. 2006). Second, it allows the combining of findings based on the analysis of primary data (e.g. data retrieved from statutes, planning instruments) with findings reported in other studies. As a result, the evaluation can capture a broader range of institutional factors affecting biodiversity protection in Queensland. Collection of primary data to support the analysis of such scale is far beyond the scope of the present study.

It is acknowledged that interviews as a source of data conventionally form part of program and policy assessments (Rossi et al. 2004, Bamberger et al. 2006). They were also considered in the initial research design. However, due to the broad scope and diversity of rules ‘mapped out’ for this study (see Figure 9 above) this source was substituted by secondary documentary sources, some of which also contain data obtained via surveys and interviews.

Most of the data is presented in descriptive form, and is arranged in several thematic clusters (e.g. power distribution system, land tenure system, environmental governance). This structure has been selected to assist understanding of the institutional system. It also allows using data for other analytical purposes and structures. Where possible, data was summarised in tables.

**Data analysis**

The analysis structures findings according to the categories of the IAD framework and is based on a qualitative approach. The performance of different sub-systems of rules is evaluated based on the nature of their influence on the ‘biodiversity protection arena’. Following the diagnostic framework (Tables 5 and 6), the analysis is structured around six major performance dimensions: (1) environmental outcomes and outputs (scope rules); (2) roles and responsibilities (position-boundary rules); (3) institutional mechanisms (choice rules); (4) environmental
information system (information rules); (5) decision-making coordination and cooperation (aggregation rules); (6) incentives and disincentives (payoff rules). Diagnostic questions guiding the analysis are incorporated in the discussion section of each chapter.

The diagnostic framework covers a broad scope of questions covering different factors of institutional design that may influence the achievement of environmental outcomes. It implies an application of a broad range of criteria from the environmental (ecological), social, economic and institutional/policy perspectives. The evaluation is based on a selected set of questions. Selection has been guided by several criteria discussed in chapter 4 (section 4.3.2), particularly:

- **Relevance** in relation to the problem and scale. The question should be relevant in the sense that it covers an important aspect of institutional environmental performance in relation to the loss of biodiversity at the state scale.

- **Availability and relevance of data.** Available sets of data need to meet criterion of causal relevance, or alternatively they should be re-interpreted in the context of biodiversity protection problem to ‘fit the lens’ of the analysis.

- **Skills and capacity.** The question should be answerable in terms of the expertise and qualification of the researcher, as well as taking into account available time and resource capacity.

Similarly to the timeframes, this study adopted a flexible approach to the selection and use of criteria and indicators for the evaluation. This can be justified for several reasons. First, the design of a comprehensive protocol suited to examine ‘real world’ governance systems is a highly challenging task. There is no ideal governance model and, consequently, no fixed sets of criteria or principles that can determine the effectiveness of a particular governance sub-system. Second, working with fixed sets of criteria derived from the literature may provide misleading outcomes. For example, devolution of regulatory roles to local governments can be interpreted as a positive factor (i.e. strengths) related to empowerment of local communities. However, it can also be a part of the state government’s ‘cost-shifting’ strategy aimed to get rid of an expense position in the budget (i.e. weakness). Similarly, availability of regulatory instruments (e.g. strategies, policies) can be used to identify the commitment of the regulatory authority to protect the natural environment. At the same time, this commitment may not be supported by adequate funding. In other words, what factors can be considered as ‘positive’ or ‘negative’ effectiveness indicators can often only be judged through linking and interpreting relevant factors.

Due to the diversity of effects produced by various sub-systems of rules the discussion broadly follows the ‘strengths and weakness’ framework. Each part identifies a range of enabling, as well as constraining, institutional factors that directly or indirectly affect ‘on ground’ biodiversity protection outcomes. They have been examined both separately and in the context of the overall configuration of rules.
5.2.4 Limitations

Discussion of the strengths and limitations of various elements of the research design of institutional analysis has been incorporated throughout this study. This section only lists the limitations specifically related to the approach employed in analysis of the Queensland Planning System and its findings. These include:

Scope of the study. The overall regulatory system is highly complex. Measures creating barriers or contributing to biodiversity protection can be contained in national and state laws and regulations, as well as in policy instruments applied at the local government level. Furthermore, they are distributed across institutional frameworks designed to address specific environmental matters, such as the use and allocation of environmental resources, nature conservation, coastal management, pollution control, biosecurity and others. The broad scope of this study, in particular matters covered in chapter 5, limits the level of detail that can be considered in the analysis. This analysis therefore concentrates on a few institutional arrangements, while recognising that there are many other factors, both institutional and non-institutional, impacting upon the environmental outcomes. For example, the role of court precedents, contribution of regional natural resource management arrangements, the regulatory role of non-governmental organisations, and the wider public are not addressed in detail.

Causal relations. Dissecting a complex institutional system into component parts, and identifying connections between the elements, is premised on a set of causality assumptions. In a ‘real life’ setting, establishing precise linkages between the sub-sets of rules and their behavioural effects is a very challenging task. As there are many intervening factors, correlations are difficult to establish and measure (Steel 2004, Breitmeier et al. 2011). Furthermore, many linkages and patterns are emergent and can be context and situation specific (Bellamy et al. 1999). Therefore, while the effort has been made to verify connections between particular sub-sets of rules and their effects on biodiversity protection outcomes, judgement bias, misinterpretation and overlooked linkages cannot be excluded.

Capabilities. Qualitative inquiry depends on the skills, training, capabilities and experience of the researcher (Patton 1990). The evaluative perspective underpinning this study developed over time. It was gained from extensive reading of a vast variety of documentary sources, participation in conferences, as well as personal communication with policy practitioners. Additional experience and insights have been gained from participation in two other research projects involving the analysis of Australian governance systems. At the same time, it is recognised that there is still an incomplete knowledge of ‘behind-the-scenes’ political processes in the Queensland government which may have affected the interpretation of some of the data.

Another aspect of capabilities is the qualification of the analyst. This analysis builds on an educational background in law, environmental management, business management and land use planning studies. Lacking in-depth practical expertise in ecology and landscape ecology
somewhat limits the analysis. For example, this study cannot evaluate in detail such aspects as the potential effects of planned land use patterns on biodiversity or suitability of landscape design to sustain species habitats in particular locations.

*Longevity of findings.* One of the objectives of the effectiveness assessment of institutional environmental performance is to allow predicting what environmental outcomes are likely to be achieved over time. To this end, the nature of institutional dynamics is a significant problem. Environmental regulatory systems continue to evolve and are changing along with the changing scale and nature of the problems. Furthermore, shifts in policy direction may depend on a mix of other drivers, such as changes in political leadership, public concerns, domestic and global markets, as well as national and international governance systems. Consequently, for how long analytical outcomes and related predictions may be valid cannot be precisely determined.

5.3 Development, land use and biodiversity conservation

The loss of biodiversity can be attributed to a broad range of human activities impacting upon the natural environment. Understanding these drivers and resulting pressures is fundamental to establishing what activities require regulation and what regulatory responses determine biodiversity protection outcomes. This section examines what are the major drivers of biodiversity loss in Australia and in the state of Queensland. It consists of three parts. The first describes the major trends in population and economy growth. The second examines the current land use pattern. The third summarises major threats to biodiversity.

5.3.1 Population and economy

Population growth and the economy, combined with changing climate conditions, are identified as the main drivers of the state of the natural environment both in Australia and in the state of Queensland (SOEC 2011, DEHP 2012).

*Population*

It is estimated that human habitation of the Australian continent began in the range of 42-48 000 years ago (Gillespie 2002). Most Indigenous Australians were hunter-gatherers with a developed, complex spiritual culture. Before European arrival, the estimated population of Indigenous Australians was 750 000 people speaking approximately 700 languages. Each clan had its own territory or ‘traditional land’ (Australian Museum 2014). The 26 January 1788, when 11 ships from Great Britain arrived to found a penal colony, is regarded as the official European settlement date.\(^\text{16}\)

As of 30 June 2013, the estimated resident population was 23 130 900 people reflecting an increase of 1.8 per cent over the previous year (ABS 2014). As of 30 June 2011, the estimated population was 22 842 300.

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\(^{16}\) Note: European Settlement is also used as a benchmark to estimate the extent of overall changes in the state of the native environment (see SOEC 2011).
Aboriginal and Torres Strait Islander population was 669,900 people or 3 per cent of the total Australian population (ABS 2013a). The Australian population is highly urbanised with over 15 million people residing in its major cities located near the coast (see Map 1) and the most prominent growth has occurred along the coastline (SOEC 2011). The average density at 30 June 2012 was 3.0 people per square kilometre (ABS 2013b), but it varies significantly across the country.

Map 1 Population distribution in Australia
Source: adapted from SOEC (2011:55)

Queensland’s settlement evolved slowly. On its separation from the colony of New South Wales in 1859, its population was estimated at 23,520 persons, accounting for 2.1 per cent of Australia’s population (excluding Indigenous people) (ABS 2009). Population expansion was of major concern during the state’s formation stage. In the early years, migration was the largest contributor to the population growth with major attractors being free passage and gifts of land orders (ABS 2009). Queensland’s population reached half a million in 1901, 1 million in 1938, 2 million in 1974, and exceeded 3 million in 1992 (ABS 2003).

Currently Queensland is the third most populated state in Australia. At 30 June 2013, the estimated population was 4,658,557 or 20.1 per cent of the national population and indicated a 2 per cent increase in last 12 months (ABS 2014). The largest contribution to population growth was made by overseas migration (50.2 per cent), followed by natural increase (39.9 per cent), and interstate migration (10.5 per cent) (ABS 2014). Queensland’s population is concentrated along the coastline with approximately 88 per cent living within 50 km of the coast (DEHP 2012). The majority resides in the South East region which accommodates around 70 per cent of the state’s population (QGSO 2012).
**Economy**

Since European settlement, Australia’s economy has undergone several shifts. Initially, the economy was oriented towards primary production with pastoral farming being the major industry. In the 1850s, with discovery of gold, Australia experienced a mining boom and expansion in banking and commerce. By 1901, the expansion had resulted in an economy with agriculture, manufacturing, mining, construction, and the service industries all providing important contributions (ABS 2012). An expansion of manufacturing industry occurred in the 1950s when its share in Gross Domestic Product (GDP) rose to around 30 per cent. The 1980s, 1990s and early 2000s saw a decline in the relative contribution to the GDP from goods-producing industries, particularly manufacturing, and a rise in the contribution from service industries (ABS 2012).

Currently, Australia has one of the largest economies in the world. In 2013, with a GDP of 1520.60 billion US dollars Australia represented 2.45 per cent of the world’s economy (Trading Economics 2012). In terms of GDP in current prices Australia is ranked 12th in the world (IMF 2014). During the last two decades there has been a consecutive growth in Australia’s economy. In the period from 1992 to 2013 its annual average real GDP growth has been 3.3 per cent (ATC 2013). A range of industries have contributed to the overall growth with the leading sectors being financial and insurance services, manufacturing, construction, and mining. Contributions of each sector and changes over the last decade are illustrated in Figure 10.

![Figure 10 Industry contributions to gross domestic product, 2001-01 and 2010-11](image)

Source: extracted from ABS (2012)

Along with major shifts in economic development, changes have also occurred in the employment structure. During the last 50 years there has been a continuous rise in employment in service industries, now reaching over 80 per cent of total employment in Australia.
According to the Reserve Bank of Australia (RBA) (2010), the fastest growing sectors have been financial and professional services, and social services such as health and education. The services sector accounts for 75 per cent of employment (see Figure 11).

![Figure 11 Trends in employment structure in Australia](image)

Consistent with national trends, Queensland’s economy evolved based on primary production. At separation from NSW, Queensland was heavily dependent on pastoral production, in particular sheep and cattle with wool accounting for 94 per cent of exports and 70 per cent of government revenue. At the end of 19th century discovery of gold and copper made the mining industry a significant part of the economy (ABS 2009).

Since the mid 1980s, except for the period from 2008-2011, the growth of Queensland’s economy, measured by Gross State Product (GSP), outperforms the average growth of the rest of Australia (QGSO 2014). From 1986 to 2013 the average annual growth rate was 4.5 per cent, compared with 3.27 per cent for Australia (QGSO 2014). In 2012-2013 Queensland’s GSP reached $290.16 billion, which accounts for 19.1 percent of Australian GDP (DFAT 2013). The major contributors to the economic growth are outlined in Figure 12.

![Figure 12 Contributions to growth in Queensland gross state product](image)

Source: extracted from Queensland Government (2013b:33)
In 2012-13 Queensland's export value was $44.44 billion (DFAT 2013). Similarly to the national economy, its exports have been dominated by commodities, mainly coal (42 per cent) and metals (8 per cent), followed by beef, cotton, wheat and vegetables (DFAT 2013). Recently, the economy has been boosted through investment in mining, in particular three liquefied natural gas projects, which are expected to contribute to further growth in gas exports (Queensland Government 2013b). Tourism is also a growing industry and in 2010-11 directly contributed $10.5 billion, and indirectly 11.4 billion, to Queensland’s GSP, or 7.8 per cent of the total GSP (Queensland Government 2014).

Employment is more evenly spread across the industries, the main ones being health care and social assistance, retail trade, and construction (see Figure 13). The major export contributors, mining and agriculture, account respectively for only 3.15 percent and 2.62 per cent of total employment. In 2012-2013 unemployment rate was 6 per cent (DFAT 2013).

Since 2012, the major strategy of the Queensland Government has been to boost economy growth in four sectors: tourism, agriculture, resources and construction. The economy is expected to reach 6 per cent GSP growth in 2015-2016, primarily driven through growth in resource export and household spending. To attract business investment, the Queensland Government has introduced so called ‘red-tape’ and ‘green-tape’ reduction strategies aimed at reducing the regulatory burden by 20% by 2018 (Queensland Government 2013b).

5.3.2 Land use pattern

Australia is the sixth largest country in the world with a total area of 7 692 024 square kilometres. The Australian mainland has a total coastline length of 35 877 km with an additional 23 859 km of island coastlines (Geoscience 2010). Queensland is situated in the north east part of the continent (see Map 1) and covers an area of 1 730 648 square kilometres (173.1 million
hectares). It is the second-largest state in Australia (23 per cent of the territory). The length of its coastline is 13 347 km.

Australia’s land use is broadly determined by climatic conditions. The largest part is arid and semi-arid, defined by the presence of desert vegetation and land forms. These so called ‘rangelands’ occupy approximately 70 per cent of Australia (Australian Government 2009). The northern part of the continent has a tropical climate, with a ‘wet’ season from approximately November through to April and a ‘dry’ season from May to October. The south-east and south-west corners have a temperate climate with variable rainfall. Australia experiences both droughts, which can last several years, and frequent flood events. Queensland lies partly in the temperate (46 per cent) and tropical (54 per cent) zones (ABS 2012).

The land is an important asset of Australia's economy. Current distribution of land uses reflects the history and pattern of European settlement (SOEC 2011). The dominant land use is livestock grazing accounting for 55 per cent of terrestrial area (see Table 8). It is predominately based on native pastures located in the rangelands, which are not suitable for other agricultural activities. Dryland cropping occurs on about 3 per cent of land, predominately in temperate and subtropical regions. Production forestry occupies 1.8 per cent, while irrigated agriculture accounts for 0.3 per cent of terrestrial area. Other uses such as urban and rural development and mining each require less than 1 per cent (ABARES 2010).

**Table 8 Land use in Australia, 2005-06**

<table>
<thead>
<tr>
<th>Land Use</th>
<th>Area million ha</th>
<th>Proportion of total area %</th>
</tr>
</thead>
<tbody>
<tr>
<td>Grazing</td>
<td></td>
<td></td>
</tr>
<tr>
<td>native vegetation</td>
<td>356</td>
<td>46%</td>
</tr>
<tr>
<td>modified pastures</td>
<td>72</td>
<td>9%</td>
</tr>
<tr>
<td>Dryland cropping</td>
<td>26</td>
<td>3%</td>
</tr>
<tr>
<td>Irrigated and intensive agriculture</td>
<td></td>
<td></td>
</tr>
<tr>
<td>irrigated cropping</td>
<td>1.3</td>
<td>0.2%</td>
</tr>
<tr>
<td>irrigated pastures</td>
<td>1.0</td>
<td>0.1%</td>
</tr>
<tr>
<td>irrigated horticulture</td>
<td>0.4</td>
<td>&lt;0.1%</td>
</tr>
<tr>
<td>intensive animal and plant production</td>
<td>0.3</td>
<td>&lt;0.1%</td>
</tr>
<tr>
<td>dryland horticulture</td>
<td>0.1</td>
<td>&lt;0.1%</td>
</tr>
<tr>
<td>Forests and plantations</td>
<td></td>
<td></td>
</tr>
<tr>
<td>native forest</td>
<td>11</td>
<td>1.5%</td>
</tr>
<tr>
<td>plantation forest</td>
<td>2</td>
<td>0.3%</td>
</tr>
<tr>
<td>Urban and rural development</td>
<td></td>
<td></td>
</tr>
<tr>
<td>intensive (mainly urban) uses</td>
<td>1.6</td>
<td>0.2%</td>
</tr>
<tr>
<td>rural residential</td>
<td>0.9</td>
<td>0.1%</td>
</tr>
<tr>
<td>Mining and waste</td>
<td>0.2</td>
<td>&lt;0.1%</td>
</tr>
<tr>
<td>Water</td>
<td>13</td>
<td>1.6%</td>
</tr>
<tr>
<td>Nature conservation and other protected areas (including Indigenous uses)</td>
<td>159</td>
<td>20%</td>
</tr>
<tr>
<td>Minimal use</td>
<td>124</td>
<td>16%</td>
</tr>
<tr>
<td><strong>Total area(b)</strong></td>
<td><strong>769</strong></td>
<td><strong>100.0</strong></td>
</tr>
</tbody>
</table>


Queensland has the largest estimated land area in agricultural use in Australia (ABS 2012). According to data reported for 2006-2007 agricultural land comprised around 80 per cent of the
The total terrestrial area of Queensland (see Table 9). The area used for crop production comprised around 2.54 million ha or 1.47 per cent, while 78.33 per cent or 135.56 million hectares is used for livestock grazing. Crop production has been concentrated along the coastal zone, as well as in the Murray-Darling basin region. Sugarcane production dominates the tropical north.

Table 9 Land use: agricultural commodities, Queensland, 2006-2007

<table>
<thead>
<tr>
<th>Land Use: Agricultural Commodities</th>
<th>Area million ha</th>
<th>Proportion of total area of Queensland %</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land under crop (including vegetables, fruits, nuts, broadacre crops, grapes and nurseries)</td>
<td>2,540,565</td>
<td>1.47%</td>
</tr>
<tr>
<td>Land under fallow</td>
<td>1,734,002</td>
<td>1%</td>
</tr>
<tr>
<td>Grazing land (including pastures and rangelands)</td>
<td>135,563,342</td>
<td>78.33%</td>
</tr>
<tr>
<td>Remnant vegetation and woodland not suitable for grazing</td>
<td>3,640,498</td>
<td>2.10%</td>
</tr>
<tr>
<td>Wetlands or swamps not suitable for grazing</td>
<td>161,703</td>
<td>0.09%</td>
</tr>
<tr>
<td>Other environmentally sensitive areas fenced out of production</td>
<td>90,919</td>
<td>0.05%</td>
</tr>
<tr>
<td>Land under commercial forestry plantations</td>
<td>224,576</td>
<td>0.13%</td>
</tr>
<tr>
<td>Houses, sheds and other agriculturally unproductive land</td>
<td>623,538</td>
<td>0.36%</td>
</tr>
<tr>
<td>Land use n.e.c.</td>
<td>268,375</td>
<td>0.16%</td>
</tr>
<tr>
<td>Not reported - area (ha)</td>
<td>675,835</td>
<td>0.39%</td>
</tr>
<tr>
<td>Total area used for agricultural commodities</td>
<td>145,523,352</td>
<td>84.09%</td>
</tr>
<tr>
<td>Total area of Queensland</td>
<td>173,064,800</td>
<td>100.0</td>
</tr>
</tbody>
</table>

Source: author based on data from ABS (2008)

Two other sectors competing for land resources are extractive industries and urban development. During the last three decades Queensland has experienced a rapid increase in the area of mining tenements. In 2011, the mining footprint (excluding coal seam gas) was 160,000 ha or 0.09 per cent of the state (DEHP 2012). Land classified as ‘extractive industries’, which includes such uses as quarries, mines, tailing dumps, well/bores, salt pans, and dredging operations, covered 657,000 hectares or 0.4 per cent of land area (ABS 2013c). Furthermore, more than half of Queensland (around 96.66 million hectares) is under some form of granted exploration or production tenure for coal, mineral, petroleum, coal seam gas or geothermal energy (DEHP 2012).

The land classified as residential accounts only for 1 per cent of the total area of Queensland. However, with population growth coastal areas experience increasing conversion of land from agriculture to uses for housing and infrastructure (DAFF 2013a). The South East Queensland region, which is the most populated area, contains 514,000 hectares or 6 per cent of the region allocated for residential use (ABS 2013c).

In Australia, the second most common use of land is nature conservation. Over the last two decades there has been a continuous increase in the area allocated for biodiversity conservation. In 2011, the National Reserve System included more than 10,000 protected areas covering over 12.7 million hectares or 16.52 per cent of the continent. It is made up of Commonwealth, state
and territory conservation reserves, Indigenous lands and private protected areas (SOEC 2011). However, these areas are not evenly distributed (see Map 2). Many of the bioregions, in particular those based on more fertile soils and the coastal areas, are underrepresented (Australian Government 2009).

![Map 2 National Reserve System in Australia, 2010-2011](image)

Source: extracted from DOE (2013)

In Queensland, protected areas are critical to the preservation of the natural environment and biodiversity. During the last decade the number of protected areas has increased. In December 2011, the total area allocated for biodiversity conservation reached 8,662,744 ha or approximately 5.01 per cent of the terrestrial area (DEPH 2012). Protected area estate established on state land has been complemented by nature refuges established on private land. In June 2011, there were 398 nature refuges covering approximately 2,799,393 ha or 1.62 per cent of the terrestrial area. However, many of the bioregions and regional ecosystems still remain underrepresented (see DEHP 2012 for a detailed review).

### 5.3.3 State of the environment and major pressures

Australia's environment is highly diverse and it is recognised as one of the 17 world’s ‘mega-diverse’ countries, with ecosystems of exceptional variety (ABS 2012). Long isolation of the continent has made many species unique. Around 80 per cent of terrestrial species of flora and fauna are endemic. About 85 per cent of inshore fish species in the southern temperate zone are found only in Australian waters. Moreover, at least 75 per cent of the native species still remain undiscovered or undescribed (Australian Government 2009).
Queensland too is rich in biologic diversity. It contains more than 12,000 plant species, 72 per cent of Australia’s native bird species, 85 per cent of its mammals, as well as over half of native reptiles and frogs (DERM 2011a). Five of Australia’s 18 World Heritage areas are fully or partially located within Queensland: the Great Barrier Reef, Wet Tropics of Queensland, Gondwana Rainforests of Australia, Fraser Island and the Australian Fossil Mammal Site (Riversleigh section). In 2009, Queensland had around 6.8 million ha of natural or near natural wetlands covering about 3.9 per cent of the state (DEHP 2012). More than 600,000 hectares are nationally significant wetlands protected under the international Ramsar Convention on Wetlands (DERM 2011a).

Australia also contains 10 per cent of the world’s threatened species (Australian Government 2009). All major national and state reports on the state of the environment identify an ongoing decline in biodiversity (OECD 2008, Australian Government 2009, SOEC 2011). Continuously reported threats to biodiversity include: (1) habitat loss, degradation and fragmentation, (2) invasive species, (3) unsustainable use and management of natural resources, (4) changes to the aquatic environment and water flows, (5) changing fire regimes and (6) changing climate (Australian Government 2009, SOEC 2011).

In Queensland, intensive use of land and environmental resources has significantly affected the state of the natural environment. In August 2011, 1372 species were listed as ‘near threatened’, ‘vulnerable’, ‘endangered’ or ‘extinct in the wild’ under the Nature Conservation (Wildlife) Regulation 2006 (Qld). The highest number of threatened vertebrate animal species and vascular plant species was reported for Southeast Queensland and Brigalow Belt bioregions, which are the regions with the highest vegetation clearing rate (DEHP 2012).

Vegetation loss remains one of the major direct causes of landscape change and biodiversity loss in Australia and in Queensland (OECD 2008, SOEC 2011, DEHP 2012). Increasing competition for land resources for human settlements, mining and agricultural production has significantly decreased the area available for biodiversity conservation. According to the State of the Environment Queensland 2011 (SOE Queensland 2011) report, Queensland had approximately 140 million hectares of remnant vegetation (81 per cent of the total area). About 71 per cent of all ecosystems contained more than 30 per cent of the pre-European vegetation, 25 per cent were described as ‘of concern’ or ‘vulnerable’ (10-30 per cent). 4 per cent of the ecosystems were ‘endangered’, with less than 10 per cent of native vegetation extent remaining (DEHP 2012). Map 3 illustrates the distribution of remnant vegetation.
A range of sectors contributing to economic growth can be causally linked to the declining state of the natural environment in Queensland. Pastoral and crop agriculture still remain the major drivers of land clearing. While broadscale clearing of vegetation was phased out in 2006, decrease of native vegetation cover continues. According to the SOE Queensland 2011, in 2008–09 woody vegetation clearing totalled 99,940 hectares per year. In the period 2005-09, the net loss of wetlands was 1890 hectares (DEHP 2012). The forestry sector also continues to compete for valuable state forest resources, and logging of native forests is expected to grow (see section 5.6).

Apart from vegetation clearing, agricultural uses produce other pressures. According to the SOE Queensland 2011, excessive grazing, in particular in vulnerable rangelands, has contributed to a decline in land and vegetation condition. It has made the land vulnerable to wind and water
erosion. The retention of livestock numbers and elevated grazing pressures during the 2001-2006 drought period adversely impacted landscape functions. Soil fertility was also declining in most of the grain cropping lands (DEHP 2012). Over the years, introduction of sheep and cattle has led to a number of species extinctions and contractions. More intensive land uses such as horticulture also involve conflicts between farmers and wildlife, such as flying foxes which feast on crops (DERM 2011a).

Agricultural practices also adversely impact upon aquatic ecosystems. Grazing has affected the condition of riparian vegetation, thus contributing to the degradation of waterways (DERM 2011a). Rural diffuse pollution is also of major concern for the health of the Great Barrier Reef, where agricultural fertiliser use is a key source of dissolved nitrogen and phosphorus run-off (DEHP 2012). Agricultural land use also competes for scarce water resources. In 2008–09, agriculture accounted for 64 per cent of Queensland’s water consumption, while households consumed only 9.1 per cent. Diversion of water streams, changes in flow regimes and waterway barriers adversely affect the habitat for freshwater species (DERM 2011a, DEHP 2012).

An ongoing national as well as state-wide threat is that of invasive species. According to the SOE Queensland 2011, Queensland had 19 mammal, 13 bird, 3 reptile, as well as 1260 naturalised plant species. Of these, 23 plant species were terrestrial ‘weeds of national significance’ (DEHP 2012). Invasive species affect biodiversity by outcompeting native species or altering essential habitats. Clearing of native vegetation and changes in vegetation cover, as a result of grazing pressures, are directly linked to the spread of invasive plant species (DEHP 2012).

In Queensland, mining occupies a comparably limited terrestrial area. However, environmental pressures related to mining and extractive industries are increasing. They are linked to site development, demands for energy and water, and pollution. Pressures on aquatic ecosystems result from discharges from mine sites, which may carry toxicants or salts, particularly during the flood events. Of particular concern is the development of the coal seam gas industry which requires a large volume of groundwater for the gas extraction process (DEHP 2012). Mining expansion is also linked to port development which has increased pressures on the Great Barrier Reef (GBR). Of recent concern has been the expansion of the Abbot Point port in north Queensland, which involves dumping of dredge material in GBR area (SKM 2013). Mining has also a significant effect on biodiversity in areas where distribution of endemic species and rare ecosystems coincide with mineral deposits (DERM 2011a).

Australia has relatively low population density. However, the concentration of population in urban centres along the coastline, and resulting urban and peri-urban expansion, threatens the biological diversity of coastal ecosystems (SOEC 2011, DEHP 2012). Human settlements affect the natural environment through land conversion for housing, industry and infrastructure development, consumption of water and energy, and the generation of waste and pollutants.
Urban and industrial development increases the demand for extractive material (e.g. sand, gravel, rocks) along the seaboard (DEHP 2012). Furthermore, development also involves the risk of disturbing acid sulfate soils, which can result in the contamination of land and coastal waters. An estimated 2.3 million ha of potential acid sulfate soils are on the Queensland coastline (DEHP 2012). Roads, powerlines, gas pipelines, and other utilities fragment habitat, and expansion of road infrastructure increases road kills of native wildlife (DERM 2011a).

In summary, biodiversity loss in Australia and the state of Queensland is an ongoing and intensifying process attributable to various uses of land resources supporting economic growth. Major losses of terrestrial biodiversity can be attributed to vegetation clearing required to support expansion of the agricultural industry, as well as human settlements in coastal areas. Vegetation clearing is most directly linked to habitat loss and fragmentation and has flow-on effects on aquatic systems, including the Great Barrier Reef.

5.4 Australian governance: foundations

The need to protect the natural environment and biodiversity are comparably recent policy problems. They are embedded in established governance systems defining distribution of regulatory and revenue raising powers and determining ‘fundamental’ rules for institutional design. To this end, many problems with ‘environmental’ regulation can be traced back to the design and operation of the overall governance system. Understanding these foundations is crucial for the analysis of any regulatory system. This section describes the institutional foundations of the Australian governance. It outlines the institutional setting determining regulatory powers and resource capacities of national, state and local governments, with particular focus on the state of Queensland. It consists of three parts. The first describes power setting at the national level. The second specifically examines the power and resource distribution system in the state of Queensland. The third outlines the foundations of statutory and strategic frameworks operating in Queensland.

5.4.1 National governance: distribution of power and revenue raising capacities

Australia has a federal system of government. The Commonwealth of Australia was formed in 1901 as a result of an agreement between six British colonies - New South Wales, Queensland, South Australia, Tasmania, Victoria, and Western Australia - which subsequently became

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17 Note: the analysis of the Queensland Planning System requires an understanding of the foundations of the system. Therefore, instead of explaining the context in each specific case a reference is made to the relevant part of this section. Readers familiar with Australian institutional foundations do not need to read this section in detail.
separate states. The Commonwealth of Australia Constitution Act 1900 (Australian Constitution) came into force on 1 January 1901. Northern Territory (NT) and Australian Capital Territory (ACT) were ceded to the Commonwealth in 1911 and received self-government rights in 1978 (NT) and 1988 (ACT). The legislatures in the two territories exercise powers delegated by the Commonwealth, but the Commonwealth Parliament retains the power to override the legislation (Australian Constitution, s122).

**Power distribution**

From Britain, Australia inherited the Westminster system of government with three separate branches: the legislature, the executive and the judiciary. Section 1 of the Australian Constitution provides for the Commonwealth Parliament, consisting of the Sovereign (the Queen), an Upper House (the Senate) and a Lower House (the House of Representatives). Legislative power of the Commonwealth is vested in the Parliament. The formal head of the Executive Government is the Governor-General as the Queen’s representative. In reality, however, the Commonwealth government is led by the Prime Minister and the Cabinet of Ministers. The Ministers are required to be elected members of one of the houses of the Commonwealth Parliament (Australian Constitution s64).

The Australian Constitution vests the judicial power of the Commonwealth in the High Court of Australia, and other federal courts that the Commonwealth Parliament creates or invests with federal jurisdiction (s71). The High Court is the judicial interpreter of the Australian Constitution, the federal legislation, as well as the final court of appeal within Australia in all other cases (Australian Constitution ss73-76). The High Court does not veto legislation, but in deciding a case it may determine that a particular enactment is unconstitutional and of no effect. The law of precedent requires that the courts adhere to previous decisions, unless they are overruled by the higher court. As a result, a law or particular provision deemed unconstitutional by the High Court becomes void and of no effect (DOHR 2012). The High Court can also overrule a state's authority in particular regulatory matters through reference to the Commonwealth Parliament's constitutional powers, as outlined below.

The distribution of legislative powers between the Commonwealth and the states is determined by the Australian Constitution. The principal legislative powers of the Commonwealth Parliament are set out in sections 51 and 52\(^\text{18}\). The overall scope of the legislative power of the Commonwealth Parliament is largely limited to the ‘heads of power’ listed in section 51. This section contains 40 sub-sections\(^\text{19}\) each describing a ‘head of power’ under which the Commonwealth Parliament is authorised to make laws. Among the most important powers are trade and commerce, corporations, taxation, postal and communication services, quarantine,

\(^{18}\) Note: section 52 determines areas within the exclusive jurisdiction of the Commonwealth Parliament, which include determining the seat of Commonwealth government, matters relating to the public service controlled by the government and any other matters declared to be within exclusive power.

\(^{19}\) Note: this includes section xxiiiA
defence, external affairs, monetary system and immigration. Powers not listed in section 51 (so
called ‘residual powers’) remain the legislative domain of the states, unless they decide to refer
a particular matter to the Commonwealth (Australian Constitution s51 (xxxvii)). This domain
includes enactment of legislation providing for natural resource management, environmental
conservation and land use planning and development (see section 5.5 for more detail). The state
can legislate on most of the matters listed in section 51. However, in case of inconsistency the
Commonwealth law prevails (Australian Constitution, s109).

Power relations between the federal and state governments have been dynamic. Their evolution
has been largely driven by four major constitutional ‘heads of power’ of the Commonwealth
Parliament: ‘interstate trade and commerce’ (s51(i)), ‘corporations’ (s51(xx)), ‘external affairs’
(s51(xxix)) and ‘taxation’ (s51(ii)). Furthermore, section 96 of the Australian Constitution
allocates to the Commonwealth Parliament the right to provide financial assistance to the states
‘on such terms and conditions as the Parliament thinks fit’. Both taxation and financial
assistance powers have given the federal government a strong position to influence the policies
of the states, and extend its powers to other matters. These powers have been also applied to
implement the federal government’s policies in the resource management sector.

In 1992, a significant change in the relations between the federal and state governments
occurred with the establishment of the Council of Australian Governments (COAG). The
COAG was established following the agreement between the Australian Prime Minister and
Premiers and Chief Ministers of the states and territories. As some analysts note, this
arrangement started the phase of ‘cooperative federalism’ in Australia (Painter 1996, Hollander
2006). Over the years, the COAG has become a core governmental forum for coordination of
powers and responsibilities between the Commonwealth and state and territory governments on
issues of national or cross-jurisdictional importance. The COAG has initiated and coordinated a
range of reform agendas, including those addressing environmental matters (see section 5.5).
Many of the COAG agreements have led to changes in national and state legislation.

Local government is the lowest tier of government in Australia. It was created in the 1840s to
enable colonial governments to deliver local services (DIRD 2013a). Local government is not
recognized in the Australian Constitution. As a result, territorial boundaries, authority, as well
as revenue raising capacity of local government are determined by the regulatory framework of
the respective state or Northern Territory. Each jurisdiction has separate local government acts
that provide the framework for the operation of this tier of government. Legislative functions of
most local government activities are undertaken by councillors elected by eligible voters.
Indigenous local governments form a distinct group in terms of roles and responsibilities, as
they may operate under different legislation (DIRD 2013a). Currently, there are 565 local
governing bodies in Australia (DIRD 2013b).

Note: ACT has only a Legislative Assembly which also performs local government functions.
There are no direct power relations between the federal and local governments. While several attempts have been made by the federal government to gain constitutional recognition of local governments, proposed amendments to the Australian Constitution were not accepted in public referenda held in 1977 and 1988. Despite the lack of formal recognition, local governments do participate in federal policy-making processes. At the national level, they are represented by the Australian Local Government Association (ALGA), which is a federation of the states’ and Northern Territory’s local government associations. This organisation represents the interests of local governments on national bodies, the COAG and ministerial councils, and provides the forum for local governments to guide national policies (ALGA 2014).

**Revenue sources and distribution**

The taxation system is one of the main determinants of power relations between the federal and state and territory governments. The major amount of national taxation income under the ‘taxation’ power is collected by the federal government. For example, in 2010 the federal government raised 80.3 per cent of Australia’s total tax revenue (The Treasury 2013). The states and territories have retained some rights to collect taxes, but their taxation income is insufficient to finance all required services. As a result, the Commonwealth grants form about 45 per cent of total state and territory government revenue (The Treasury 2013).

The current fiscal relations between the Commonwealth and the states and territories are based on two major reforms. In 2000, the federal government introduced the Goods and Service Tax (GST), which is a value added tax of 10 per cent levied on most transactions with goods and services. While centrally collected, all of the GST revenues are distributed by the Commonwealth Grants Commission (CGC) to the states and territories as unconditional (untied) grants. The GST distribution follows the principle of horizontal fiscal equalisation and is updated annually to reflect changes in the circumstances of the states (CGC 2012).

In 2008, the federal, state and territory governments signed the *Intergovernmental Agreement on Federal Financial Relations* (IAFFR), which aimed to reduce Commonwealth prescriptions on service delivery by the states and territories. The IAFFR provided further clarification of fiscal relations. According to the Agreement (clause 19) the federal government committed to provide ongoing financial support for the states’ and territories’ service delivery through four major instruments:

(a) general revenue assistance, including the on-going provision of GST payments, to be used by the states and territories for any purpose;
(b) National Specific Purpose Payments (NSPPs) to be spent in the key service delivery sectors;
(c) National Health Reform Funding; and
(d) National Partnership payments (NPPs) to support the delivery of specified outputs or projects, to facilitate reforms or to reward those jurisdictions that deliver on nationally significant reforms.

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21 Note: the aim of the principle is to offset interstate differences in revenue raising capacity and the costs of providing services and acquiring infrastructure (CGC 2012)
The IAFFR includes separate agreements appended as schedules which specify particular financial arrangements.

Local governments have their own sources of revenue. The most part comes from property rates, which are the only tax instrument, and the fees and charges for provided goods and services (DIRD 2013a). In 2010-2011 local government taxation revenue amounted to 3.5 per cent of all taxes raised across all levels of government (DIRD 2013a). Local governments vary considerably in their revenue raising capacity (DIRD 2013a). According to the review of the Productivity Commission (2008), urban local governments were predominately funded from their own sources, whereas in most rural and remote areas the Commonwealth and state/territory grants formed a substantial part (44 per cent and more) of the revenue. The problem of declining financial sustainability of local governments has been raised at the national level on several occasions (see SCEFPA 2003 for identified problems).

The Australian Constitution does not provide the Commonwealth with the power to directly fund local governments. However, since the 1970s the federal government provides regular financial assistance to support local governments in the form of general purpose and special purpose (local roads) grants. Currently, this assistance is regulated under the Local Government (Financial Assistance) Act 1995 (Cth). Grants are paid to state and territory governments, which pass them to the local governments, based on the recommendations of local government grants commissions (DIRD 2013a). The federal government periodically provides other funding to support particular local government initiatives.

5.4.2 Queensland: distribution of power and revenue raising capacities

Queensland, as a separate British colony, was formed in 1859 by its separation from the colony of New South Wales. The Constitution Act 1867 (Qld) (Constitution Act) provided the basis for the operation of the Queensland Parliament and its regulatory powers. In 2001, the state of Queensland adopted a new codified constitution, repealing as well as consolidating several Acts of Parliament comprising constitutional law. The Constitution of Queensland Act 2001 (Qld) (COQA) came into effect on 6 June 2002.

Power distribution

Similarly to other states, Queensland’s system of government follows the Westminster model. The Constitution Act authorises the Parliament of Queensland to make laws ‘for the peace, welfare and good government’ of Queensland (s2). The Parliament consists of the Queen, represented by the Governor, and the Legislative Assembly (Constitution Act s2A). Queensland is the only jurisdiction in Australia which has a unicameral Parliament or Legislative Assembly.22 The electoral cycle is three years (Constitution Act s2) and voting is compulsory.

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22 Note: upper house or the Queensland Legislative Council was abolished in 1922.
The Assembly can legislate on any matters subject to the limits set out in the Australian Constitution (see section 5.4.1 above).

The Cabinet is the principal decision-making body of the executive government and it is collectively responsible to the Parliament (COQA s42). It consists of the Premier and the Ministers. The Ministers are appointed by the Governor on the Premier’s recommendation. The maximum number of Ministers that can be appointed by the Governor is 19 (COQA s43(4)). It is a common practice that Ministers are elected members of the Legislative Assembly. Successful operation of the Cabinet requires the majority support of the Legislative Assembly.

The principal responsibilities of the Ministers are set out in administrative arrangements made by the Order in Council. The arrangements declare legislation portfolios and respective administrative units (departments or their parts) administered by each Minister (COQA s44). The ultimate responsibility for governmental performance within the scope of an allocated portfolio rests with the Minister. In practice, however, every portfolio is managed by government departments or their sub-units, which develop and implement policies and undertake required management and monitoring functions. The management of departments is the responsibility of a Director-General who is the Chief Executive Officer (DPC 2013). As of March 2012, the Queensland government consists of 19 appointed Ministers (including the Premier) having responsibility for a corresponding number of government departments.

The COQA establishes a judicial system which consists of the Supreme Court and the District Court (s57). The Supreme Court is the highest court. Subject to the provisions of the Australian Constitution, it has unlimited jurisdiction in Queensland (COQA s58). There are other courts and tribunals established under different Acts of Parliament (e.g. the Planning and Environment Court, the Land Court). Queensland’s constitutional laws do not establish a rigid separation of powers. The Legislative Assembly can override the decisions of the Supreme Court. The Assembly however is bound by decisions of the High Court of Australia (see section 5.4.1).

Local government as a separate tier of government is recognised in the COQA and is described as ‘an elected body that is charged with the good rule and local government of a part of Queensland allocated to the body’ (s71). Constitution, responsibilities and powers of local governments are established under the Local Government Act 2009 (Qld) (LGA), which is the central statute providing for the local government system in Queensland. A separate City of Brisbane Act 2010 (Qld) provides for the constitution and operation of the Brisbane City Council. The term of office of a local government is four years and participation in elections is mandatory. Since 2009 Indigenous local governments are regulated under the LGA which contains special provisions addressing their operation.

The operation of local governments is regulated and controlled by the Queensland Government. Under the LGA the responsible Minister holds the authority on behalf of the state to monitor and evaluate performance of the local government and its councillors, revoke or suspend
decisions and local laws, initiate change in local government area, name or representation, remove a councillor, or dissolve a local government (ss18, 38AB, 90B, 113, 120-124). Dissolution of local government must be ratified in the Legislative Assembly (COQA s72). The state government can also delegate (devolve) performance of particular regulatory functions to local governments under other Acts of Parliament.

The overall scope of authority of local governments in Queensland is established in the LGA. The Act provides for the power of local government ‘to do anything that is necessary or convenient for the good rule and local government of its local government area’ provided that such actions are within the scope of authority of the state government (s9). Apart from some specified functions (e.g. responsibility for local roads), the LGA provides the local governments with a considerable degree of flexibility regarding the choice and delivery of the goods and services to communities.

The LGA also allocates to local government the power to make and enforce a local law for its local government area (s28(1)). However, local laws must consider the overall state interest (s29A) and they cannot be inconsistent with state law (s27). The Minister can also approve ‘a model local law’ suitable for incorporation by all local governments into their local laws (s26(7)). In 2013, there were approximately 3000 local laws in Queensland (DLGCRR 2013b).

The Local Government Association of Queensland (LGAQ) is the peak body representing the interests of Queensland local governments at state and national levels. The LGAQ performs an advocacy role in the policy-making process. It also assists local governments in the delivery of their services through the established procurement system and consulting services (LGAQ 2012). Major themes of the state–local government ‘power debate’ are delineation of state and local interests, local government autonomy and funding (see LGAQ 2013).

During the last two decades, the local government system in Queensland has undergone several reforms. In 2007, in response to financial problems, the Queensland government initiated a major structural reform of local government areas. The reform took effect on 15 March 2008 and resulted in compulsory amalgamations of many local governments and reduced the number from 157 to 73. In 2013 four councils voted in support of de-amalgamation. As a result, since January 2014 there are 77 local governments: 7 cities (including Brisbane), 40 shires and 30 regions. Of these 14 local government areas are classified as Aboriginal shire councils and 2 as Indigenous regional councils.

Revenue sources and distribution

The funding of services provided by the Queensland government is based on several revenue streams. In 2012-2013 Commonwealth grants (GST revenue, NSPPs and NPPs) formed 45 per cent of the total income, while taxation was the major own source revenue accounting for 26 per cent. The biggest contributors to the ‘other revenue’ group were mining royalties accounting for
5.5 per cent of total revenues (see Figure 14, left chart). Key taxation sources were payroll tax, stamp duties levied on property transactions, motor vehicle registration, gambling and land tax (see Figure 14, right chart) (Queensland Government 2013b). In competition with other states and territories the Queensland government follows a low taxation strategy to attract business investments (Queensland Government 2013b).

**General government revenue**

**Taxation revenue**

<table>
<thead>
<tr>
<th>Source</th>
<th>Revenue</th>
<th>Percentage</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sales of goods and services</td>
<td>$5,087 m</td>
<td>12%</td>
</tr>
<tr>
<td>Grants revenue</td>
<td>$18,295 m</td>
<td>45%</td>
</tr>
<tr>
<td>Interest income</td>
<td>$2,603 m</td>
<td>6%</td>
</tr>
<tr>
<td>Dividend and income tax</td>
<td>$1,390 m</td>
<td>3%</td>
</tr>
<tr>
<td>Other revenue</td>
<td>$3,413 m</td>
<td>8%</td>
</tr>
<tr>
<td>Taxation revenue</td>
<td>$10,957 m</td>
<td>26%</td>
</tr>
<tr>
<td>Land taxes</td>
<td>$990 m</td>
<td>9%</td>
</tr>
<tr>
<td>Taxes on employers</td>
<td>$3,751 m</td>
<td>35%</td>
</tr>
<tr>
<td>Stamp duties</td>
<td>$1,887 m</td>
<td>17%</td>
</tr>
<tr>
<td>Other</td>
<td>$630 m</td>
<td>6%</td>
</tr>
<tr>
<td>Taxes on gambling</td>
<td>$1,034 m</td>
<td>9%</td>
</tr>
<tr>
<td>Taxes on insurance</td>
<td>$670 m</td>
<td>6%</td>
</tr>
<tr>
<td>Motor vehicle taxes</td>
<td>$1,995 m</td>
<td>18%</td>
</tr>
<tr>
<td>Taxes on employers</td>
<td>$3,751 m</td>
<td>35%</td>
</tr>
</tbody>
</table>

**Figure 14** Composition of general government revenues and taxation revenues in Queensland, 2012-2013
Source: adapted from Queensland Government (2013b:49, 61)

The Queensland Government provides a broad range of services. According to the 2013-2014 budget data (Queensland Government 2013b) the main expense positions were health and education forming around 50 per cent of the overall government expenses (see Figure 15). For the social sector, federal government Health reform payments and other SPPs (schools, skills, workforce development, disability services and affordable housing) form a source of funding.

**Figure 15** Government expenses by purpose, Queensland 2013-2014 projections
Source: adapted from Queensland Government (2013b:87)

As budgeted for 2013-14, general government expenses are $48.436 billion with an estimated fiscal deficit of $7.664 billion. Since 2007-2008, the Queensland government has operated the budget with increasing fiscal deficit. It has risen from $995 million in 2007-2008 to a peak of $8.686 billion in 2012-2013. Projected government debt for 2014-15 is estimated at $85 billion (Queensland Government 2013b).
Since March 2012 the Queensland government has implemented fiscal repair reform, focused on a reduction of government expenditure with planned restoration of income-expense balance and stabilisation of general government debt. Part of the strategy involves reduction of the size of the public service, which in 2012-2013 reduced 12800 full time equivalent positions in government departments (Queensland Government 2013b). This reform has also affected government spending for environmental conservation.

Queensland’s local governments have a separate source of revenue. There are six main sources of revenue available to local governments: (1) rates and charges, (2) fees, (3) profits from council owned businesses, (4) grants and subsidies, (5) loans and (6) developer contributions and infrastructure charges levied on land being developed (DLGCRR 2013c). In 2010-2011, on average local government taxation (rates) constituted 30.3 per cent of local government revenue (DIDR 2013a).

Queensland’s local governments differ significantly in their revenue raising capacities. For example, in 2010-2011 the rates and charges in Cairns Regional Council constituted 78.43 per cent, and in Fraser Coast Regional Council 73.37 per cent, of operating income. In contrast, Barcoo Shire Council secured only 3.27 per cent. Overall, 21 councils (out of 59 councils reporting financial data) received less than 20 per cent of their operating income from rates and charges (see DLGCRR 2013a for financial information). Consequently, grants and subsidies play a significant role in financial sustainability of non-urban and remote local governments, and determine the extent of affordable services.

5.4.3 Queensland: regulatory foundations

The regulatory framework operating in the state Queensland has evolved over time in response to different political, social, economic and environmental triggers. At the same time, irrespective of the regulatory matter concerned (e.g. environment), its design follows some commonly accepted principles, structure and established hierarchy as outlined below.

**Statutory foundations**

Acts of Parliament are the major regulatory mechanism providing for rights and obligations of Queensland residents. In general, the Queensland Parliament has a plenary (i.e. full) power to make laws for the state of Queensland. However, according to the Australian Constitution, certain Commonwealth Acts of Parliament (e.g. taxation laws) apply also in Queensland. Commonwealth legislation can also confer jurisdiction on states with regard to certain matters (e.g. Coastal Waters (State Powers) Act 1980 (Cth)). The federal government can also delegate certain regulatory functions performed under the Commonwealth legislation to the Queensland government.

There are no strict prescriptions as to what matters should be regulated under the Acts of Parliament or how the Acts should be designed. However, it is commonly recommended that an
implementation of a government policy through the legislation should be considered in those cases which involve modification of existing rights and obligations, or require enforcement measures, permanency or is of high importance (DPC 2014). Basically, the legislative scheme consists of directly applicable codes of conduct (behavioural prescriptions), and rules prescribing the way the legislation operates through other laws, instruments and decisions. The design depends on the matter, convenience, practicality and appropriateness of delegated powers (DPC 2014). Currently, three Acts of Parliament form the foundation for the design and operation of laws and other regulatory instruments in Queensland: the Legislative Standards Act 1992 (Qld) (LSA), the Acts Interpretation Act 1954 (Qld) (AIA) and the Statutory Instruments Act 1992 (Qld) (SIA).

The LSA sets out the framework of fundamental legislative principles. According to section 4(1), fundamental legislative principles are ‘the principles relating to legislation that underlie a parliamentary democracy based on the rule of law’. They include the requirement that legislation has sufficient regard to the rights and liberties of individuals and the institution of Parliament (LSA s4(2)). The LSA also specifies some of the matters that should be considered in determining whether legislation observes these principles. Among such matters, section 4(3) includes: consistency with principles of natural justice, avoidance of provisions that adversely and retrospectively affects rights or liberties, or retrospectively imposes obligations and fair compensation for the compulsory acquisition of property (see LSA for the comprehensive list).

The LSA also has a mandatory requirement to provide explanatory notes for bills (i.e. draft acts) and significant subordinate legislation (s22). Among other matters, the explanatory notes are required to include a brief assessment of the consistency of the legislation with fundamental legislative principles, and a statement of the reasons for any inconsistency (ss23(1)(f), 24(1)(i)). Inconsistency requires strong justification (DPC 2014). The LSA establishes the Office of the Queensland Parliamentary Counsel as having responsibility to provide advice in relation to the application of fundamental legislative principles in drafting legislation (s7).

The AIA assists ‘in the shortening and interpretation of Queensland Acts’. This Act provides for a range of rules of statutory construction and interpretation, and contains provisions that apply to all Queensland legislation (ss2,6). For example, the AIA establishes that a reference to a law ‘includes a reference to the statutory instruments made or in force under the law or provision’ (s7(1)), or that the interpretation of the provision ‘that will best achieve the purpose of the Act is to be preferred to any other interpretation’ (s14A). The AIA also specifies the terms of commencement, amendment and repeal of Acts (Part 5 and 6), the functions and powers conferred by Acts (Part 7), terms and references in Acts (Part 8), as well as the meaning of commonly used words and expressions (Schedule 1). The application of the AIA may be ‘displaced, wholly or partly, by a contrary intention appearing in any Act’ (s4). Interpretation and analysis of statutes and other statutory instruments requires being familiar with these provisions.
Acts of Parliament may directly regulate a particular matter, or authorise another decision-making body (e.g. Governor in Council, Minister, chief executive, local government) to regulate or decide the matter. The instruments created by exercising these powers tend to be commonly referred to as ‘delegated legislation’. Development, publication and interpretation of these instruments are regulated under the SIA, which describes a statutory instrument as an instrument that is made under an Act or another statutory instrument, and is of a particular type (see s7(3) for a list). Subordinate legislation, a sub-category of statutory instruments, must be tabled in the Legislative Assembly and can be disallowed by it (s49).

The SIA also specifies the extent of regulation-making powers. Section 22 provides that, if an Act authorises or requires the making of a statutory instrument, the power includes a power to make a statutory instrument with respect to any matter that ‘is required or permitted to be prescribed by the authorising law or other law or is necessary or convenient to be prescribed for carrying out or giving effect to the authorising law or other law’. Statutory instruments must not exceed the power conferred by the law under which it is made (s21). As a general rule, an instrument can only provide for beneficial retrospectivity, which means it cannot retrospectively decrease a person’s rights or impose liabilities on a person (s34). Similarly, the application of the SIA may be displaced by ‘stating contrary intention’ (s4).

Along with other states, Queensland has inherited the English system of ‘common law’, or law originating in decisions made by judges. In the statutory context, understanding the role of common law is important for several reasons. First, the legislation is written against the background of common law and incorporates established principles. Many of the fundamental legislative principles originate in common law (DPC 2014). Second, many matters relating to private rights (e.g. land use), not specifically regulated in statutes, will be decided in accordance with common law. The general approach taken by a court to interpretation of statutory rights is a presumption that a statute does not intend to affect the common law rights, unless by express or necessary intendment those rights have been modified or displaced (Bates 2003). For example, a narrow common law interpretation of legal standing (e.g. the right to commence court action) may impede the rights of the public to challenge government decisions in public interest, unless the Act does not specify legal standing requirements (Bates 2003). Finally, the judges undertake interpretation of statutory provisions and may specify how a particular provision is to be understood.

**Strategic foundations**

Statutes and common law form only a part of the regulatory framework. While often not considered by legal analysts, so called ‘strategic’ frameworks form an important part of the regulatory system. Since the end of the 1980s, the Queensland Government, along with other Australian governments, introduced public sector reforms focused on cost reduction, efficiency and administrative accountability. These reforms, often labelled as ‘new managerialism’ (Hood
introduced a broad range of management techniques derived from the private sector. Among others, they led to the introduction of new sets of rules, such as vision and mission statements, strategic goals and objectives, action plans and performance indicators guiding regulatory authorities. While in most cases legally non-enforceable, they play significant roles in policy-making and implementation.

The organisational structure of the Queensland Government is complex, with different structural units and sub-units managing multiple matters, and interacting with diverse stakeholder groups (see section 5.5.2 for environmental governance). Actions of regulatory authorities are guided by numerous strategies, policies, guidelines, targets, models and supporting tools. There are no formal hierarchies of strategies, and the extent of their application (i.e. ‘behavioural significance’) often remains unclear. For this reason, this study is guided by the ‘Value Chain Model’ of the Performance Management Framework (see Figure 16). This model was introduced by the Department of the Premier and Cabinet (DPC) (2012a) to clarify linkages between the parts of the Queensland public sector and underpinning strategic frameworks.

Figure 16 The Value Chain Model for the Queensland Public Sector
Source: extracted from DPC (2012a:4)

According to the model, at the top of the hierarchy are so called ‘objectives for the community’, which largely reflect election commitments, and the program of the leading political party with majority seats in the Legislative Assembly. The objectives and supporting actions are further detailed in a set of ‘whole of government’ strategies, policies, plans and frameworks. Each agency such as department is guided by its own strategic framework, which includes vision, mission and objective statements, and performance targets aligned with the overall government priorities and election commitments. This framework underpins annual reports published by
each agency (see Queensland Government 2013a for a list). This framework is further complemented by strategies, policies and guidelines for agency-stakeholder interactions in particular management sectors, matters or localities.

The role of the ‘strategic framework’ should not be underestimated. Many provisions (e.g. departmental goals and objectives) drive the policy-making process and, as will be outlined in sections 5.5 and 5.6, precede changes in statutory systems. Furthermore, many strategic frameworks designed to resolve specific policy problems are supported by action plans and funding mechanisms and play an important role in guiding behaviour of targeted groups of actors. Despite unenforceability, these rules may have higher behavioural significance than some of the Acts of the Parliament. In examining such ‘whole of government’ problems as biodiversity loss, they cannot be excluded from the analysis.

5.5 Environmental governance

This section examines the distribution of regulatory authorities among the federal, state and local governments and the institutional framework regulating the use of land and environmental resources in the state of Queensland and consists of two parts. The first examines regulatory power, capacities and environmental regulation operating at the national level. The second details current distribution of environmental roles and responsibilities in the state of Queensland and the operating strategic and regulatory frameworks.

5.5.1 National environmental governance

Distribution of roles and responsibilities

Environmental problems were not of particular concern in the federation building period. The Australian Constitution does not incorporate provisions stating obligation to protect the natural environment. Under the Constitution the Commonwealth Parliament has limited jurisdiction over the use and distribution of environmental resources. Section 51 allocates only one ‘head of power’ to the Commonwealth Parliament, which concerns ‘fisheries in Australian waters beyond territorial limits’ (s51(x)). However, several indirect ‘heads of power’ have given the Commonwealth considerable influence in environmental matters.

Of particular importance has been the ‘external affairs’ power which enables the Commonwealth Parliament to pass legislation to implement obligations under international agreements. Australia is a contracting party to many international agreements addressing environmental problems. They include: the Convention on Biological Diversity 1992 (Biodiversity Convention), the United Nations Framework Convention on Climate Change 1992 and its Kyoto Protocol, Convention on International Trade in Endangered Species of Wild Fauna and Flora 1973, Convention on Wetlands of International Importance especially as Waterfowl Habitat 1971, Convention concerning the Protection of the World Cultural and

In the 1970s several jurisdictional disputes emerged with regard to the use of marine resources. In line with UNCLOS, in 1973 the Commonwealth Parliament passed the Seas and Submerged Lands Act 1973 (Cth), which established Commonwealth jurisdiction over all territorial waters, sea bed and air space. The following High Court decision in New South Wales v. Commonwealth (1976) 135 CLR 337 upheld the right of the Commonwealth Parliament to legislate over the territorial sea. In 1979 the Commonwealth and the states arrived at the so-called 'Offshore Constitutional Settlement' determining the jurisdiction over marine resources. In 1980 the Commonwealth Parliament passed the Coastal Waters (State Powers) Act 1980 (Cth), which extended the legislative powers of the states in relation to coastal waters from the low water mark to three nautical miles offshore. Commonwealth jurisdiction, if not specified otherwise, has been set from the three nautical mile limit to the 200 nautical mile limit of the Australian Fishing Zone.

Since the 1990s the Commonwealth and the states have adopted a more cooperative approach in resolving jurisdictional disputes in environmental matters (Bates 2003). Significant changes in intergovernmental relations occurred with the establishment of the COAG (see section 5.4.1). In 1992 the COAG reached a major agreement setting the foundation for the current environmental governance in Australia. The Intergovernmental Agreement on the Environment (IGAE) was concluded between the federal and all state and territory governments and the ALGA as a representative of local governments. It aimed to define ‘the roles, responsibilities and interests of all levels of the Government in relation to the environment’ (clause 2.1.1).

According to the IGAE the federal government retains the responsibility for management of environmental resources on Commonwealth land and for ‘national environmental matters’, which according to clause 2.2.1 include:

1. matters of foreign policy relating to the environment and, in particular, negotiating and entering into international agreements relating to the environment and ensuring that international obligations relating to the environment are met by Australia;
2. ensuring that the policies or practices of a State do not result in significant adverse external effects in relation to the environment of another State or the lands or territories of the Commonwealth or maritime areas within Australia's jurisdiction (subject to any existing Commonwealth legislative arrangements in relation to maritime areas);
3. facilitating the co-operative development of national environmental standards and guidelines as agreed in Schedules to this Agreement.

The states and territories retain full responsibility ‘for the development and implementation of policy in relation to environmental matters which have no significant effects on matters which are the responsibility of the Commonwealth or any other State’ (clause 2.3.1).
The IGAE also provided the framework for environmental decision-making, policy development and implementation. It introduced ‘ecologically sustainable development’ as the major principle of environmental policy and programs, and specified four guiding principles: the precautionary principle, the principle of inter-generational equity, the principle of conservation of biological diversity and ecological integrity, and the principle of improved valuation, pricing and incentive mechanisms (section 3). The IGAE contained nine schedules addressing specific areas of environmental policy and management: (1) data collection and handling; (2) resource assessment, land use decisions and approval processes; (3) environmental impact assessment; (4) national environment protection measures; (5) climate change; (6) biological diversity; (7) national estate; (8) world heritage; and (9) nature conservation. Schedule 9 explicitly recognised that ‘the States have primary responsibility in the general area of nature conservation’ (clause 2).

At the national level, there is no common agreement regarding what environmental matters should be regulated at the local government level. Clause 2.4.1 of the IGAE established the responsibility of local governments for ‘the development and implementation of locally relevant and applicable environmental policies within its jurisdiction in co-operation with other levels of Government and the local community’. The Agreement also acknowledged that local governments ‘have an interest in the environment of their localities and in the environments to which they are linked’ (clause 2.4.2). Supporting schedules, however, did not provide detailed description of rights and responsibilities of this tier of government.

**Statutory and strategic framework**


More recently, Commonwealth regulation has been extended to other matters. The *Water Act 2007 (Cth)* provides for centralised water resource planning and allocation in the Murray-Darling Basin. It evolved in response to the water over-allocation problem and resulting environmental degradation in the basin.\(^{23}\) The *National Greenhouse and Energy Reporting Act*

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\(^{23}\) Note: for environmental degradation history in the Murray-Darling Basin and the conflicts leading to Commonwealth control see Connell (2007).
2007 (Cth) established a national reporting system for large greenhouse gas emitters. One of the scheme’s objectives is to create the framework for carbon trading mechanisms. In 2011 the Commonwealth Parliament enacted the *Clean Energy Act 2011* (Cth), introducing a package of legislation establishing an emissions trading scheme (commonly known as the ‘carbon tax’). This mechanism, however, has been subject to political debate and the recently elected Coalition federal government has promised to repeal the Act.

In 1999 the Commonwealth Parliament enacted the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) which is the centrepiece of national environmental legislation. The EPBC Act defines environmental matters of Commonwealth concern, including matters of national environmental significance. It provides for the identification of threatening processes, establishes a process of environmental impact assessment, and provides an approval system for projects having a significant impact on national environmental matters. As of 2013, there are nine matters of national environmental significance:

- world heritage properties,
- national heritage places,
- wetlands of international importance (i.e. 'Ramsar wetlands’),
- nationally threatened species and ecological communities,
- migratory species,
- Commonwealth marine areas,
- the Great Barrier Reef Marine Park,
- nuclear actions (including uranium mining),
- water resources in relation to coal seam gas and large coal mining development.

The EPBC Act (s44) allows the federal government to conclude bilateral agreements with the states and territories delegating assessment and approval powers. Assessment bilateral agreements allow the assessment of ‘controlled action’ under the accredited state/territory environmental impact assessment processes. The approval, however, still remains the responsibility of the Minister of the federal government. Approval under the bilateral agreement enables the state/territory governments to assess and approve the action (or not) under the provisions of the EPBC Act. Currently, all states and territories have concluded assessment bilateral agreements. However, two draft approval agreements with New South Wales and Queensland have been opened for public comment (deadline 13.07.2014).

An important part of national environmental policy is state of environment (SOE) reporting. Introduced by the *National Strategy for Ecologically Sustainable Development*, provisions for the SOE report have been incorporated in the EPBC Act. The Act requires the Minister to prepare and table before the Commonwealth Parliament ‘a report on the environment in the Australian jurisdiction’ every 5 years (s516B). The fundamental objectives of the SEO are to provide information to support decisions on environmental policies, and to give the public access to information on the state of the Australian environment. The report is prepared by an independent committee and its scope is determined by the responsible Commonwealth department (DOE 2014). Four reports (1996, 2001, 2006 and 2011) have been released. In general, they follow some variation of ‘pressure-state-response’ framework identifying the
current condition of the environment and its resources, the pressures determining the condition, and implemented management responses and their impacts (DOE 2014). Most of the states and territories prepare their own SOEs.

In addition to the statutory framework, numerous national policies and strategies have been endorsed by the COAG on important environmental matters. In 1992, along with the IGAE, the second most significant agreement was the National Strategy for Ecologically Sustainable Development (the National ESD Strategy) (ESDSC 1992). It was adopted in response to the 1987 report of the World Commission on Environment and Development. The National ESD Strategy (Part 1) defined the concept of ‘ecologically sustainable development’ (ESD) as:

using, conserving and enhancing the community’s resources so that ecological processes, on which life depends, are maintained, and the total quality of life, now and in the future, can be increased.

To support implementation, it introduced a set of ESD principles, including the precautionary principle, the principles of intergenerational equity, conservation of biological diversity and ecological integrity, and improved valuation, pricing and incentive mechanisms.

The National ESD Strategy (Part 2) provided for a list of objectives and actions for key industry sectors relying on natural resources as their productive base. The implementation, however, was optional. It was recognised that ‘in light of the very significant budgetary constraints’ each jurisdiction will determine its own priorities for implementation (Part 1). Over the years most of the action plans have lost their significance (Ross and Dovers 2006). At the same time, the ESD has become an important overarching objective and value criterion of resource management policies, and many statutes incorporate variations of established principles (Bates 2003). Both the IGAE and the National ESD Strategy have established the foundation for a range of other national environmental policies and strategies.

Australia ratified the Biodiversity Convention in June 1993. In 1996 the Commonwealth and all states and territories endorsed the National Strategy for the Conservation of Australia’s Biological Diversity (National Biodiversity Strategy 1996) aimed ‘to protect biological diversity and maintain ecological processes and systems’ (ANZECC 1996:11). The strategy was the main mechanism for implementation of Australia’s obligations under the Convention. Five years later it was complemented by the National Objectives and Targets for Biodiversity Conservation 2001–2005 (DEH 2001). The targets, however, were not agreed by all state and territory governments and consistent implementation was lacking (Australian Government 2009).

In 2010 the National Biodiversity Strategy 1996 was replaced by Australia’s Biodiversity Conservation Strategy 2010–2030 (National Biodiversity Strategy 2010). The new strategy is designed to ‘guide all Australian biodiversity strategies and policies, including those of the Australian, state and territory governments and the private sector, that address specific aspects of biodiversity conservation’ (NRMMC 2010:37). It contains three priorities for actions: (1) engaging all Australians in biodiversity conservation; (2) building ecosystem resilience in a changing climate; and (3) getting measurable results. Each priority is further detailed in three
sub-priorities: outcomes, targets and actions. The Strategy is supported by 10 interim national targets for 2015 (NRMMC 2010). Four outcomes (2.1.1-2.14) are established for the sub-priority ‘protecting diversity’ (NRMMC 2010:44):

1. an increase in the number, extent and condition of ecosystems protected under secure conservation tenure;
2. an increase in the extent of private land managed for biodiversity conservation;
3. an improvement in the conservation status of listed threatened species and ecological communities;
4. a net national increase in the extent and condition of native habitat across tenures.

The 1990s brought a new era in land use and resource management policies. In response to different triggers the federal government and the COAG have introduced a range of policies, strategies and frameworks, aiming to develop uniform approaches to national environmental problems, such as environmental pollution and sustainable management of forest, fisheries, water and soil resources, vegetation cover and coastal zone. Implementation of national policies and agreements has been supported by various programs and Commonwealth funding. While not formally enforceable, they have triggered changes in the legislation and environmental policies of the states and territories.

An overall strategic framework is extensive and cannot be examined in detail within the scope of this chapter.²⁴ Five strategic frameworks are most directly linked to the National Biodiversity Strategy 2010: National Framework for the Management and Monitoring of Australia’s Native Vegetation (SCEW 2012), Australia’s Strategy for the National Reserve System 2009-2030 (NRMMC 2009), Australian Weeds Strategy (NRMMC 2006) and Australian Pest Animal Strategy (NRMMC 2007). Other important frameworks are the National Forest Policy Statement 1992, which was signed by the federal, state and territory governments to resolve forest logging issues, and the National Water Initiative 2004, which aims to introduce a nationally compatible water planning and market system, to resolve water overallocation and overuse problem and to improve environmental outcomes (NWC 2014). The reform has been implemented under the supervision of the National Water Commission established under the National Water Commission Act 2004 (Cth).

Organisational and funding arrangements

A range of administrative bodies have been created to facilitate consultation and cooperation between the Commonwealth and the states and territories. The COAG is supported by the ministerial council system. Historically, environmental and resource management policies have been coordinated through several councils, such as the National Environment Protection Council and the Australian and New Zealand Environmental Conservation Council, later replaced by the Resource Management Ministerial Council. This organisational structure is dynamic and has undergone several reforms. On 13 December 2013 the COAG introduced

²⁴ Note: for more extensive list of strategies see Australian Government (2009)
another major reform by replacing its 22 Standing Councils with eight Councils to ‘refocus on COAG’s priorities over the next 12 to 18 months’ (COAG 2014). Under the current structure all ‘environmental’ councils have been abolished and the immediate future of national environmental policy issues remains unclear.

Distribution of funding with regard to environmental matters is not fixed. The Intergovernmental Agreement on Federal Financial Relations (see section 5.4.1) does not provide for ‘environmental’ Special Purpose Payments. In general, state/territory governments are responsible for funding allocation for the implementation of environmental legislation, policies and programs, as well as for acquisition and management of protected areas on public lands such as national parks and state forests (Australian Government 2009). The federal government’s expenditure relates to funding of environmental programs in support of national strategies and agreements, which can be transferred via National Purpose Payments. The federal government is also the initiator and funder of research. It supports several research organisations including the Commonwealth Scientific and Industrial Research Organisation (CSIRO), the Australian Institute of Marine Science, Geoscience Australia, and the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES). National data services are also provided through the Bureau of Meteorology (BOM).

In the last two decades the most extensive funding commitment of the federal government has been the Natural Heritage Trust (NHT). In 1997 the Commonwealth Parliament enacted the Natural Heritage Trust of Australia Act 1997 (Cth) (NHTA Act) with the main objective to establish the account ‘to conserve, repair and replenish Australia’s natural capital infrastructure’ (s3). A funding program for the Natural Heritage Trust (NHT) was created under the NHTA Act. In the period from 1997 to 2008, the NHT provided funding of $3.1 billion for projects that restore and conserve Australia’s environment and natural resources. It was allocated among a range of programs such as: National Landcare Program, Farm Forestry Program, National Rivercare Initiative, Murray-Darling 2001 Initiative, Endangered Species Program, National Reserve System Program (SOEC 2011). In 2008, the NHT was consolidated into funding the program Caring for Our Country with a budget of $2.25 billion over a five year period (2008–2013) (SOEC 2011). This program continues with the federal government committing to resources of more than $2 billion for the next three years. Delivery is secured through two streams: Sustainable Agriculture and Sustainable Environment (Australian Government 2013).

Implementation of national environmental programs has also led to the creation of new governance actors, such as community-based regional natural resource management bodies. A regional approach was introduced nationally under two programs: Nature Heritage Trust and

Note: at the state level regional approach to resource planning and funding distribution was implemented beginning 1990’s under Integrated Catchment Management umbrella to address land (soil) and water degradation issues (see Ewing 2003 for a history). Community (primary producer) led management bodies were part of the delivery of the national Landcare program first introduced in 1989.
the National Action Plan for Salinity and Water Quality. In the period from 2002-2004 federal, state and territory governments agreed on boundaries of 56 natural resource management (NRM) regions. Their operation and status was determined through bilateral agreements concluded between the federal government and respective state or territory governments. Resourcing NRM activities is a shared responsibility and conditioned upon matching funds. For example, the 2004 Bilateral Agreement between the Commonwealth of Australia and the State of Queensland requires equal allocation of funding for regional activities (clauses 118-124).

Under the NHT, regional management committees received a range of responsibilities to deliver NRM outcomes, through regional planning and partnership building, data gathering, monitoring and coordination of on-ground work (HC Coombs Policy Forum 2011). The established NRM bodies differ across jurisdictions. In some jurisdictions they have a statutory authority. For example, in NSW introduction of the NRM framework was supported by a legislative reform. The Catchment Management Authorities Act 2003 (NSW) established 13 Catchment Management Authorities (CMAs). Similarly, statutory NRM boards (Catchment Management Authorities) operate in Victoria under the Catchment and Land Protection Act 1994 (Vic). In contrast, NRM bodies in Queensland and Western Australia do not have statutory authority.

Unlike governments, regional management bodies do not have separate revenue raising powers to support implementation of their plans. Their operation is dependent on the funding from the federal, state and territory governments and other contributing sources. To this end, their operation has been volatile with several shifts in direction dictated by the funding body (see HC Coombs Policy Forum 2011 for a review).

Another national policy having potential to contribute to biodiversity protection outcomes is the Carbon Farming Initiative (CFI) which commenced in December 2011 under the Carbon Credits (Carbon Farming Initiative) Act 2011 (Cth). The CFI is a government supported voluntary carbon offset scheme aimed to increase carbon stored in vegetation and soil. It enables landholders to earn revenues from greenhouse gas abatement projects. Revenue is created by selling carbon credits (Australian Carbon Credit Units) to a buyer in the carbon market (DERM 2011b). This initiative, however, is in an early stage of evolution and could be affected by the federal government’s intention to abolish the carbon tax.

5.5.2 Environmental governance in Queensland

The environmental regulations operating in Queensland are complex. The Queensland Parliament has enacted over 30 Acts of Parliament regulating various activities impacting upon the natural environment (for a comprehensive review see McGrath 2011). The Acts are complemented by an extensive set of statutory and non-statutory instruments, as well as strategic frameworks setting out objectives for particular regulatory and administrative activities. This section examines only the major organisational and institutional arrangements operating as determinants of biodiversity protection outcomes.
**Distribution of regulatory roles and responsibilities**

In Queensland, regulatory powers with regard to environmental matters are concentrated at the state government level. Legislative portfolios are distributed among several Ministers and responsible government departments carrying out delegated regulatory and management functions (see section 5.4.2). The administrative structure is dynamic. Natural resource management and environmental conservation portfolios are amalgamated, divided and redistributed on a regular (i.e. election cycle) basis. The current administrative structure (as of 2012-2013) and distribution of environmental legislation portfolios is described in Table 10.

**Table 10** Distribution of environmental portfolios, Queensland Government 2012-2013

<table>
<thead>
<tr>
<th>Major functions</th>
<th>Environmental legislation portfolio*</th>
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<tbody>
<tr>
<td><strong>Department of Environment and Heritage Protection (DEHP)</strong></td>
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</table>
| Responsibility for environment and heritage protection, including: | Cape York Peninsula Heritage Act 2007  
Coastal Protection and Management Act 1995  
Environmental Protection Act 1994  
Mineral Resources Act 1989  
(to the extent that it is relevant to environmental matters)  
Nature Conservation Act 1992  
(except to demonstrated and exhibited native animals and management of the national and regional parks and forest reserves)  
Waste Reduction and Recycling Act 2011  
Water Act 2000  
(Chapter 3 Underground water management)  
Wet Tropics World Heritage Protection and Management Act 1993  
Wild Rivers Act 2005 |
| - preservation of the diversity and integrity of Queensland’s natural ecosystems; | |
| - development and coordination of the policy, planning and legislative frameworks for environmental and heritage protection; | |
| - management and monitoring of environmental risks, environmental impact assessments, compliance investigation and enforcement. | |
| DEHP is the coordinating agency in Queensland for the administration of the EPBC Act. | |
| **Department of Natural Resources and Mines (DNRM)** | |
| Responsibility for distribution and management of Queensland’s land and natural resources. Four functional areas include: | Aboriginal Land Act 1991  
Cape York Peninsula Heritage Act 2007  
(partially administered by DEHP)  
Geothermal Energy Act 2010  
(partially administered by Treasurer and Minister for Trade)  
Greenhouse Gas Storage Act 2009  
Land Act 1994  
Land Title Act 1994  
Land Protection (Pest and Stock Route) Management Act 2002  
(to the extent relevant to stock route management)  
Native Title (Queensland) Act 1993  
Mineral Resources Act 1989  
(partially administered by the Treasurer and Minister for Trade and DEHP)  
Petroleum and Gas (Production and Safety) Act 2004  
(except to the extent administered by the Treasurer and Minister for Trade)  
Survey and Mapping Infrastructure Act 2003  
Strategic Cropping Land Act 2011  
Torres Strait Islander Land Act 1991  
Vegetation Management Act 1999  
Water Act 2000  
(except to the extent administered by the DEHP and the Department for Energy and Water Supply) |
| - mining and petroleum services; | |
| - mine safety and health services; | |
| - water services; | |
| - land services. | |
| The DNRM is also responsible for information systems registering resource allocation and distribution and land, vegetation and resource mapping systems. SmartMap Information Services (SMIS) provide a single point of access to spatially located land information. | |
### Department of Agriculture, Fisheries and Forestry (DAFF)

Responsibility for development of **agriculture**, **fisheries** and **forestry** sector and food industry. Functional areas include:
- agricultural research, development and extension;
- forestry research and management;
- fisheries research and management;
- management of biological, animal welfare and product integrity risks.

<table>
<thead>
<tr>
<th>Statute</th>
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<tbody>
<tr>
<td><strong>Forestry Act 1959</strong> (jointly administered with the DNPRSR)</td>
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<tr>
<td><strong>Fisheries Act 1994</strong> (except for fish habitat areas)</td>
</tr>
<tr>
<td><strong>Land Protection (Pest and Stock Route) Management Act 2002</strong> (except to the extent that it is relevant to stock route management)</td>
</tr>
<tr>
<td><strong>Nature Conservation Act 1992</strong> (to the extent relevant for demonstrated and exhibited native animals)</td>
</tr>
<tr>
<td><strong>Plant Protection Act 1989</strong></td>
</tr>
<tr>
<td><strong>Strategic Cropping Land Act 2011</strong> (Chapter 5 excluding sections 139(1), 143 and 144)</td>
</tr>
<tr>
<td><strong>Torres Strait Fisheries Act 1984</strong></td>
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</tbody>
</table>

### Department of National Parks, Recreation, Sport and Racing (DNPRSR)

Responsibility for management of national parks and forests, recreational and sporting activities and racing industry.

Queensland Parks and Wildlife Service (QPWS) is a part of the DNPRSR.

<table>
<thead>
<tr>
<th>Statute</th>
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<tbody>
<tr>
<td><strong>Fisheries Act 1994</strong> (as it relates to fish habitat areas)</td>
</tr>
<tr>
<td><strong>Forestry Act 1959</strong> (as it relates to forest reserves)</td>
</tr>
<tr>
<td><strong>Marine Protection Act 2004</strong></td>
</tr>
<tr>
<td><strong>Nature Conservation Act 1992</strong> (to the extent relevant to the management of national and regional parks and forest reserves, excluding nature refuges)</td>
</tr>
<tr>
<td><strong>Recreational Areas Management Act 2006</strong></td>
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### Department of State Development, Infrastructure and Planning (DSDIP)

Responsibility for economic and regional development, major project delivery, government land and asset management, and infrastructure and planning.

Part of the DSDIP is the **State Assessment and Referral Agency (SARA)**. Since 1.07.2013 SARA coordinates development assessment process where the State government has a jurisdiction as a final decision-maker. SARA is a final decision-maker in the development assessment process.

<table>
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<th>Statute</th>
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<tr>
<td><strong>Economic Development Act 2012</strong></td>
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<td><strong>State Development and Public Works Organisation Act 1971</strong></td>
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<tr>
<td><strong>Sustainable Planning Act 2009</strong></td>
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### Department of Science, Information Technology, Innovation and the Arts

Responsibility for information and knowledge provision to support government functions, decision-making and policy development.

**Science Delivery** unit provides scientific information and advice to inform planning and management processes across the environment and natural resources spectrum.

The **Queensland Herbarium** is the centre for research and information on the Queensland flora, vegetation and plant communities. It undertakes vegetation survey and regional ecosystem mapping used for conservation and land use assessments.

*This Table lists major statutes providing for planning, distribution and use of the environmental and land resources, statutes described in more detail are marked bold (for the full list of statutes administered by each department see the sources listed below)*


In addition, there are several bodies having specific regulatory and advisory functions. For example, a separate Wet Tropics Management Authority has been established for the Wet
Tropics World Heritage area under the *Wet Tropics World Heritage Protection and Management Act 1993* (Qld). Water authorities (49 in total) have been established under the *Water Act 2000* (Qld) to carry out water activities, such as water supply for agricultural and domestic purposes and provision of coordinated drainage systems (DNRM 2013a). Departments also coordinate several advisory boards and committees providing policy or scientific advice on specific policy matters (see e.g. DEHP 2013 for a list).

Queensland local governments perform a range of regulatory functions delegated by the state government. They have substantial regulatory authority in several environmental matters, which include land use planning and development assessment under the *Sustainable Development Act 2009* (Qld) and *Coastal Protection and Management Act 1995* (Qld), licensing of some environmentally relevant activities under the *Environmental Protection Act 1994* (Qld), pest and weed control under the *Land Protection (Pest & Stock Route Management) Act 2002* (Qld) (Productivity Commission 2012). Most recently in 2013, management responsibilities for several regional (conservation) parks established under the *Nature Conservation (Protected Areas Management) Regulation 2006* (Qld) were transferred to several local governments. The statutes also prescribe the fees local governments can collect for performance of specified regulatory function.

In addition to delegated regulatory functions, Queensland local governments are authorised to adopt local laws regulating environmental matters. Such regulation is enabled under section 27 of the *Local Government Act 2009* (Qld) (LGA). The local governments are also authorised to apply incentive measures and collect levies to promote biodiversity conservation (see section 5.6.4 for details).

The regional NRM system (see section 5.5.1) consists of 13 community-based regional NRM bodies and the Torres Strait Regional Authority. These groups are responsible for the delivery of NRM plans and their coordination and implementation activities. Their work is coordinated through the Department of Natural Resources and Mines. The current focus of investment programs is on ‘on ground’ activities undertaking to protect and restore waterways, address weeds and pests and improve soil and vegetation quality (DNRM 2014a). The NRM bodies do not have a formal regulatory role. However, some elements of the regional NRM plans have been recognised in state planning instruments, namely in the *South East Queensland Regional Plan* (see chapter 6 for more detailed description of this instrument).

**Strategic framework**

Since the March 2012 elections, the central focus of the LNP Queensland government has been economic growth based on ‘four pillars’: agriculture, resources, construction and tourism (Queensland Government 2013b). The government’s interest in boosting growth in these sectors underpins numerous initiatives, including reforms in established strategic and statutory frameworks. Many of the previously adopted strategies were subjected to major revisions, with
some of them remaining as ‘rules on paper’ lacking further support. Therefore, to give an overall direction, this section outlines some of the previous arrangements and recent changes.

Over the last two decades, there has been a gradual increase in the area of state land reserved for nature conservation in Queensland. One of the relatively recent achievements leading to increase in protected area estate was the *South East Queensland Forests Agreement* (SEQ Forest Agreement), which formed part of a larger forest policy framework of the federal government. The SEQ Forest Agreement was concluded in 1999 between the Queensland government, the Australian Rainforest Conservation Society, the Queensland Conservation Council, the Wilderness Society and the Queensland Timber Board. The Agreement provided the timber industry with 25-year sales permits guaranteeing resource supply, upon condition of phasing-out logging of state native forests in SEQ by 2025 (Jarred 2000). The immediate benefit of the Agreement was the allocation of 425 000 hectares of native forests to the conservation reserve, with an additional 17 per cent categorised as ‘a last resort for logging’. While regulated under the *Forestry Act 1959* (Qld), forest land declared as forest reserve is also managed for biodiversity conservation and, subsequently, can be protected under national park tenures (see section 5.6 for explanation of tenures).

The Delbessie Agreement, also known as the *State Rural Leasehold Land Strategy* was another significant turn towards improvement of land and vegetation management practices on Crown leasehold land (see section 5.6). Finalising 10 years of consultation process, the Agreement was signed in December 2007 between the Queensland government, the Australian Rainforest Conservation Society and the representative of agricultural industries AgForce. The Delbessie Agreement established the framework for regulatory reform. It provided for a mixture of incentives and legal remedies for landholders to protect land resources under the ‘duty of care’ provisions (the *Land Act 1994* (Qld) s199) and, if required, to engage in conservation of environmental values. The framework also specified criteria for the assessment of land conditions and introduced land management agreements. ‘Good land condition’ was set out as one of the measures for prolonged lease terms (see section 5.6 for more detail). Government funds were allocated to assist landholders with development of land management plans and improvement of land management practices (DEHP 2012).

In 1996 the Queensland government committed to the implementation of the National Biodiversity Strategy 1996. In 2011, following endorsement of the new National Biodiversity Strategy, a separate ‘whole of government’ strategy was adopted by the previous Labor government. *Building Nature’s Resilience: Queensland Biodiversity Strategy* (Queensland Biodiversity Strategy) incorporated two primary goals to be achieved by 2020: reverse the decline in biodiversity, and increase the resilience of species, ecosystems and ecological processes. The strategy included commitment to the expansion of the national park estate to 7.5 per cent, or 13 million hectares, with the total protected area being 20 million hectares. The
goals also confirmed the government’s commitment to continue transfer of state forests into forest reserves (DERM 2011a).

The recent political change to the LNP government has significantly affected the established strategic framework, as the government has stepped back from several commitments. It has scaled back the Delbessie Agreement, announcing that it will not be imposing requirements to prepare land management agreements on large grazing properties, and will not require conservation agreements as a condition of extended lease periods (DNRM 2014b). Similarly, no commitments to achieve stated biodiversity conservation targets can be found in recent government reports. Moreover, references to the Queensland Biodiversity Strategy, as well as the document itself, have ‘disappeared’ from the websites of responsible agencies.

Further policy direction in the ‘environmental’ sector, at least for the period 2012-2015, is indicated in several strategic frameworks. On the highest level (see section 5.4.3 for hierarchy), the government’s Getting Queensland back on track: Statement of objectives for the community (DPC 2012b) sets out two ‘environmental’ commitments:

- improving management of our National Parks including controlling weeds and pests, and provide real protection for our dugongs, turtles, and koalas;
- supporting our community, sporting, volunteer, and local environment groups through targeted grants.

Progress reports (Queensland Government 2013c) indicate that these commitments are to be achieved through several measures. ‘Real protection’ for dugongs and turtles is achieved through the regulatory amendments with regard to illegal trade in dugong and turtle meat and products. Koala protection measures involve investment in the four year period of ‘up to’ $26.5 million in total for acquiring koala habitat, research into preventable causes of death and koala rescue and rehabilitation services. Other financial commitments include: ‘up to’ $107 million support for initiatives providing ‘real protection’ for natural heritage, $37.7 million for protected area estate acquisitions and the Nature Assist program and $12 million for Everyone’s Environment program for community-based environmental initiatives (see Queensland Government 2013c for details).

Another important ‘whole of government’ strategic framework is the draft Queensland Plan: a 30-year vision for Queensland (Queensland Government 2013d) prepared by the Queensland government in a public consultation process. The Plan intends to set out a high level direction for Queensland, which is further implemented through supporting strategies and action plans. The final version of the Plan is intended to be approved in mid-2014. Currently (as of May 2014), the draft sets out nine foundation areas of interest, which are supported by vision statements, high level long term targets, goals, outcomes, success indicators, and sets of success measures. ‘Environment—achieving balance’ is recognised as one of the foundational areas. Table 11 outlines four outcomes sought for an overarching ‘we protect the environment’ goal (goal 21) and their success descriptors (emphasis added).
### Table 11 Environmental protection outcomes, draft Queensland Plan 2013

<table>
<thead>
<tr>
<th>Outcomes</th>
<th>Success descriptors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unique environments are protected and well maintained</td>
<td>We ensure <strong>National Parks, World Heritage Areas</strong>, prime agricultural land and <strong>significant ecosystems</strong> are maintained and <strong>support economic growth</strong>.</td>
</tr>
<tr>
<td>Our natural environment has economic value</td>
<td><strong>We have a healthy natural environment</strong> that is a <strong>key contributor to our economy.</strong> <strong>We maintain a natural environment that underpins tourism and key industries such as agriculture.</strong></td>
</tr>
<tr>
<td>Decisions are based on scientific evidence</td>
<td><strong>We undertake research, invest in and implement internationally renowned alternative energy solutions.</strong> <strong>We develop industry, community and government partnerships to preserve Queensland’s landscapes and wildlife.</strong></td>
</tr>
<tr>
<td>Environmental education encourages personal responsibility</td>
<td><strong>We include environmental education in learning and development programs.</strong> <strong>We support communities to care for local natural environments.</strong> <strong>We manage our personal consumption of natural resources more wisely.</strong></td>
</tr>
</tbody>
</table>

Source: information extracted from Queensland Government (2013e:20)

*Queensland’s Agricultural Strategy* (DAFF 2013b) is another strategic framework having significant implications for the future state of the natural environment. Introduced in June 2013 the Strategy sets out an ambitious vision for agriculture, fisheries and forestry: to double the agricultural production by 2040. The Strategy identifies four key pathways (DAFF 2013b):

- securing and increasing resource availability,
- driving productivity growth across the supply chain,
- securing and increasing market access, and
- minimising the cost of production.

The need to increase availability of productive land resources is further linked to changes in several regulatory frameworks, which will increase pressures on the natural environment. First, recent reforms in the *Vegetation Management Act 1999* (Qld) (VMA) have enabled clearing of remnant vegetation for high value agricultural purposes (see chapter 6 for details). Second, the Government has committed to increase water availability by releases of unallocated water for agricultural purposes and increases in water storages. Third, a new measure is ‘drought emergency grazing’ provisions introduced through reforms in the *Nature Conservation Act 1992* (Qld) (see below). They allow temporary grazing permits to be issued in several national parks (current provisions applied until 31.12.2013). Finally, stated commitments for the forestry sector (DAFF 2013b) suggest that there will be an increase in state forest area reserved for timber industry at the expense of the conservation reserve.

At a more general level, the LNP government’s target is to reduce the regulatory burden on businesses and the community. As part of the reform, it has created The Office of Best Practice Regulation which is responsible for the regulatory reform. Implementation of so called ‘red-tape’ and ‘green-tape’ reduction implies removal of some regulatory control measures placing restrictions on market access. Under this strategy significant reforms in environmental approval processes have been made in the *Environmental Protection Act 1994* (Qld) (see DEHP 2013 for details).
Statutory framework

Queensland’s ‘environmental’ statutes have evolved over a long period of time, often in an ‘ad hoc’ and experimental fashion. Incorporated regulatory scope and applied drafting techniques do not allow structuring them into distinct and separate groups (see McGrath 2010 for a discussion). However, for the ease of review, this section describes the overall system by using three broad descriptive categories: (1) frameworks providing for resource distribution and planning; (2) frameworks regulating multiple uses of extractive and land resources in specified areas; and (3) frameworks regulating harmful activities.

In Queensland, the ownership rights to the major groups of extractive environmental resources (e.g. water, fish, minerals, petroleum, native animals) are vested in the state. The regulation of resource allocation and planning follows a ‘sectoral’ pattern with separate statutory frameworks designed for each type of the resource. A water planning and allocation system is established under the *Water Act 2000* (Qld), fisheries resources are regulated under the *Fisheries Act 1994* (Qld), wildlife under the *Nature Conservation Act 1992* (Qld) and forest resources (including wildlife in state forests) under the *Forestry Act 1959* (Qld). Allocation and extraction of mineral resources is regulated under the *Mineral Resources Act 1989* (Qld), petroleum products under the *Petroleum and Gas (Production and Safety) Act 2004* (Qld) and geothermal energy under the *Geothermal Energy Act 2010* (Qld) (see section 5.6.3 for mining tenements). The *Land Act 1994* (Qld), which regulates allocation and use of the Crown leasehold land, is examined in section 5.6 of this chapter.

Statutes providing for the allocation and distribution of extractive environmental resources differ in the regulatory scope and approaches to resource allocation and planning. However, as Bates (2003) summarises, they contain some common features:

- allocation of regulatory authority and associated rights and responsibilities (e.g. responsible Minister),
- creation of new authorities responsible for particular functions (e.g. planning committees, advisory committees, panels, tribunals);
- resource planning, which may include provisions specifying type and content of the management plan, planning and approval process, rights and responsibilities of involved actors;
- resource distribution system, which includes provisions for resource access authorities (e.g. licences, permits) and their application conditions (e.g. resource extraction limit, timing, area, amount, use of technology), rights associated with the resource authority (e.g. transfer of quotas, compensation) and related charges;
- prohibited and restricted activities (e.g. mining in conservation areas), which adversely affect the resource and require a permit from the regulator (e.g. damaging forest resources or marine plants, dredging);
- management of other processes that adversely affect the resource (e.g. management of weed and pest species, land erosion);
- provisions for offences, criminal and civil sanctions and enforcement proceedings;
- conflict resolution mechanisms and processes.

Multiple uses of land and environmental resources in specified areas are regulated under several statutes. At one end of the spectrum are statutes providing for the management of areas with a dominant purpose to protect environmental values or a particular resource (e.g. national parks, marine parks). At the other end are statutes providing for planning and development of areas which require ‘balancing’ protection of environmental values against competing and conflicting land uses.

In the area of environmental protection the Nature Protection Act 1992 (Qld) (NCA) is the most significant piece of legislation providing for terrestrial biodiversity conservation in Queensland. The NCA and its subordinate legislation determine the declaration, administration, planning, and permissible uses of the state land reserves set aside for biodiversity conservation, as well as the establishment and management of private protected areas (see section 5.6.4 for permissible scope of activities). The NCA provides for five classes of protected wildlife (s71), and establishes classification criteria and management principles (ss76-80). The Nature Conservation (Wildlife) Regulation 2006 (Qld) lists the protected species.

The Marine Parks Act 2004 (Qld) provides for the declaration, zoning and management of marine protected areas. These areas are complemented by protected fish habitat areas declared under the Fisheries Act 1994 (Qld)26.

The Wild Rivers Act 2005 (Qld) (WRA) adds another layer of protected areas, providing for integrated catchment management of unmodified rivers. The WRA and subordinate legislation restrict, as well as prohibit some land-, water-, vegetation- and extractive mineral and petroleum resource-related activities that could significantly affect protected attributes. Ten declarations have been made under the WRA. However, some of them, such as rivers in Cape York, are planned to be revoked by the LNP government to support expansion of mining activities in the area (Queensland Government 2013c). The development assessment process of the WRA is carried out under the SPA.

The Forestry Act 1959 (Qld) (Forestry Act) establishes a separate ‘integrated’ land and resource use framework for state forests set aside for production of timber and associated products (s33). It regulates the declaration, management and permissible uses (e.g. grazing, recreational activities, getting of quarry material) of state forest land. Both the NCA and the Forestry Act

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26 Note: There are 70 declared fish habitat areas covering 1,134,288 ha of tidal land in Queensland (DEHP 2012).
regulate the use of the state reserve land and associated soil, vegetation and wildlife resources, but do not cover water allocation or quality issues.

The *Coastal Protection and Management Act 1995* (Qld) (CPMA) provides for the delineation of the coastal zone, and for a separate planning and development control regime to protect coastal resources. It also provides for the regulation of quarrying, canal construction and other tidal works in the coastal zone and coastal management districts. Major regulatory instruments of the CPMA (state planning policies, development assessment codes) are integrated in the planning and development assessment system established under the SPA (see below).

The *Sustainable Planning Act 2009* (Qld) (SPA) is the major regulatory framework providing for development activities other than mining, petroleum and state-scale infrastructure projects. Its overarching purpose is ‘ecological sustainability’ to be achieved through planning instruments and integrated development assessment system (IDAS). The term ‘development’ is defined broadly, integrating assessment of various activities in the common framework. The SPA also incorporates the environmental impact assessment process accredited under the EPBC Act (see section 5.5.1.2). The SPA incorporates two levels of planning instruments: state and local (for a detailed description of the system see chapter 6).

Apart from the SPA, there are other state-level planning systems regulating land development. The *State Development and Public Works Organisation Act 1971* (Qld) (SDPWOA) allocates a range of powers to the state government to facilitate large-scale infrastructure and other development projects. The SDPWOA includes such instruments as declarations of state development areas, coordinated projects and prescribed development, environmental impact assessment (accredited under the EPBC Act), and a power to compulsorily acquire land. As of May 2014, there are eight state development areas in Queensland (DSDIP 2014b).

The *Economic Development Act 2012* (Qld) provides for particular parts of the state to be declared as priority development areas. It establishes a new authority (Minister for Economic Development Queensland) to plan, carry out, promote and coordinate the development of land in these areas. As of November 2013, there are 23 priority development areas regulated under the Act (DSDIP 2013b).

Finally, separate planning and development assessment frameworks have been established for road and transport infrastructure. Planning, construction and management of roads of national and state significance, rail, bus, air and marine transport infrastructure is regulated under the *Transport Infrastructure Act 1994* (Qld). The *Transport Planning and Coordination Act 1994* (Qld) provides for strategic planning and management of transport resources. Management of the stock route network is provided under the *Land Protection (Pest and Stock Route) Management Act 2002* (Qld).27

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27 Note: the stock route network is a part of state infrastructure required to support movement of cattle and sheep.
The *Environmental Protection Act 1994* (Qld) (EPA) is a central component of the environmental legislation in Queensland, and it regulates activities having potential to cause environmental harm. The overall framework consists of a complex set of regulatory instruments. It incorporates separate policies on water, air, noise and waste management, and provides environmental quality standards and indicators. A core component is the list of ‘environmentally relevant activities’ (ERAs) which require assessment under the EPA. The *Environmental Protection Regulation 2008* (Qld) lists 64 ERAs (Schedule 2). The EPA sets out a system of environmental impact assessment for mining activities (EPBC Act accredited), evaluations and audits, programs that manage environmentally affected areas and resources, as well as regulations for environmental offences.

In the light of extensive use of land resources for agricultural purposes, vegetation clearing in Australia has become another ‘harmful’ activity adversely affecting the quality of the natural environment. In Queensland, the *Vegetation Management Act 1999* (Qld) (VMA) was enacted to limit vegetation clearing on freehold land, and subsequently also on Crown leasehold tenures. The Act applies to clearing of all woody vegetation. Clearing is defined as ‘development’ under the SPA and is assessed based on policies, codes and maps developed under the VMA. This framework will be examined in detail in chapter 6.

Vegetation clearing restrictions are imposed under several other statutes. The NCA provides for another layer of protection for essential habitats of protected species. The *Fisheries Act 1994* (Qld) provides for the protection of the coastal vegetation by prohibiting to ‘remove, destroy or damage’ plants growing on tidal land (s123). This restriction applies across all tenures. Some development activities (e.g. excavation works or placing a fill) adversely affecting riverine vegetation require a permit under the *Water Act 2000* (Qld). Development assessment of activities which affect protected, marine and riverine vegetation, as well as declared fish habitat areas, are assessed under the SPA.

Finally, the *Land Protection (Pest and Stock Route) Management Act 2002* (Qld) sets out regulations for the control of declared pests. Schedule 2 of the *Land Protection (Pest & Stock Route Management) Regulations 2003* (Qld) lists declared pests in three classes, based on their current or potential economic, environmental or social impact.

Integration and alignment of the diverse requirements established under different regulatory frameworks has been an ongoing problem for the Queensland government. One of the major achievements in this regard was the introduction of an integrated development assessment system (IDAS) established under the *Integrated Planning Act 1997* (Qld), a predecessor of the SPA. In practice, employed drafting techniques do not allow viewing many Queensland’s statutes as separate systems, because they are linked through a referral system and through common statutory instruments designed to achieve the purposes of several statutes. For example, mining leases require obtaining an environmental authority under the EPA. The
recently introduced *State Planning Policy* (DSDIP 2013c), an instrument established under the SPA, incorporates provisions to assist the achievement of the objectives of other Acts (e.g. CPMA, EPA and NCA). During the recent reforms, many separate instruments (e.g. Queensland Coastal Plan) have been abolished to create a simplified system (see chapter 6).

Queensland’s local governments add little to the overall regulatory framework. The VMA ‘does not prevent a local law from imposing requirements on the clearing of vegetation in its local government area’ (s7(2)). However, as the recent review (conducted in February 2014) of the local law database reveals, only 7 non-indigenous local governments continued to regulate vegetation management and 10 had separate provisions for local parks (see DLGCRR 2013b). Most councils have introduced the model law of the state government, addressing such environmental issues as declaration and control of local pests, overgrown allotments, fire and fire hazard, community safety hazards and noise.

### 5.6 Land tenure system

The land ownership system and associated rights and responsibilities are the backbone of any land use planning system. In the environmental context, it determines the scope of permissible actions each landholder can undertake in dealing with vegetation and soil resources. While this system forms part of an overall regulatory framework operating in the state of Queensland, it requires a more detailed understanding.

This section consists of four parts. The first describes the overall land tenure system in Queensland. The second examines evolution of Indigenous land use rights. The third describes the nature of mining tenements. Finally, the section concludes with a brief review of regulatory instruments established to support biodiversity conservation on public and private land.

#### 5.6.1 Land tenures

At the time of European settlement in Australia the doctrine of ‘terra nullius’ (i.e. ‘land belonging to no one’) was applied. Following the British feudal tenure system, land ownership was vested in the Crown. Over the years, the Crown (i.e. the governments representing the Crown) alienated the land by grants and orders to the settlers (ABS 2012). As a result, two major types of private land tenures have evolved: freehold land and Crown leasehold land. Unallocated land and land set apart for public purposes is managed by the respective tier of government (federal, state or local) in the public interest. Until the 1970s Indigenous Australians were not acknowledged as rightful land owners and did not have land rights. Acknowledgment of Indigenous rights and the diversification of land uses and associated rights have created a complex system of tenures which is briefly described below (for a comprehensive analysis see Bradbrook et al. 2011).
Freehold title is the most secure form of land ownership in Australia. In freehold tenure the landholder holds the title and possession of the land. The landholder is entitled to use the land in any manner subject to restrictions and obligations imposed by the Crown grant, common law and/or legislation (Bates 2003). Owners have the right to sell, transfer or mortgage the land and exclude others from its use. Subject to restrictions, freehold land can be used for different purposes, such as agricultural and pastoral production, forest, residential, business and industrial use (Geoscience 2011). Granted interest in land is not absolute. In all states and territories, the Crown (the government) retains the interest in minerals and other mining resources that lie on and under the surface of the land (see section 5.6.3 for details).

In leasehold tenure the landholder has the right to use (possess) the land, but ownership is retained by the Crown. Leasehold tenure originated in the pastoral leasehold system introduced by the colonial governments. During the 19th and 20th centuries it became the dominant tenure of land used for pastoral production (ABS 2012). Leases generally restrict the landholder’s right to use the land for other purposes than those allocated. In addition to the interest in minerals and other mining resources, certain rights associated with the leasehold land may remain the property of the Crown. In Queensland, the Forestry Act 1959 (Qld) declares all forest products and quarry material on Crown holdings (including term lease, perpetual lease, occupational licence or permit) to be the property of the Crown (s45). Subject to stated exceptions, the Act prohibits any interference with trees or other forest products, otherwise than in accordance with a permit, lease, licence or other granted authority (s53). Subject to approval from the Crown (government), a leaseholder can sell the lease, sublet, subdivide or amalgamate the land.

While common law provides for general principles of land ownership, each state and territory has developed statutory systems regulating land tenures. In Queensland, the tenure system is regulated under the Land Title Act 1994 (Qld) (LTA) and Land Act 1994 (Qld) (Land Act). The LTA provides for registration of freehold title. Administration, management and transfer of non-freehold or Crown land are regulated under the Land Act. In leasehold tenure lessees must comply with the purpose and conditions of the lease agreement, as well as the provisions of the Land Act, which provides for several types of tenures of non-freehold land, and contains provisions regarding land allocation, administration, use, terms of holding and transfer of land rights. Current distribution of land resources in Queensland, their legal status (tenure type) and total value is provided in Table 12.
### Table 12 Land tenures in Queensland, 2013

<table>
<thead>
<tr>
<th>Tenure Type</th>
<th>Number</th>
<th>Total Area of Land (ha)</th>
<th>% of the State</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pastoral holding</td>
<td>1,148</td>
<td>72,708,484</td>
<td>41.94%</td>
<td>17,465,725,001</td>
</tr>
<tr>
<td>Occupation licence</td>
<td>159</td>
<td>636,642</td>
<td>0.37%</td>
<td>244,170,001</td>
</tr>
<tr>
<td>Grazing homestead perpetual lease</td>
<td>2,678</td>
<td>20,038,481</td>
<td>11.56%</td>
<td>17,367,855,001</td>
</tr>
<tr>
<td>Non-competitive lease</td>
<td>395</td>
<td>16,762</td>
<td>0.01%</td>
<td>2,416,604,731</td>
</tr>
<tr>
<td>Special lease</td>
<td>2,150</td>
<td>1,579,262</td>
<td>0.91%</td>
<td>3,938,089,991</td>
</tr>
<tr>
<td>Development lease</td>
<td>2</td>
<td>476</td>
<td>0.00%</td>
<td>61,000,001</td>
</tr>
<tr>
<td>Permit to occupy</td>
<td>4,425</td>
<td>583,800</td>
<td>0.34%</td>
<td>1,371,425,281</td>
</tr>
<tr>
<td>Road licence</td>
<td>4,357</td>
<td>29,033</td>
<td>0.02%</td>
<td>421,647,801</td>
</tr>
<tr>
<td>Term lease</td>
<td>5,994</td>
<td>14,892,640</td>
<td>8.59%</td>
<td>12,720,101,011</td>
</tr>
<tr>
<td>Perpetual lease</td>
<td>208</td>
<td>165,574</td>
<td>0.10%</td>
<td>2,260,838,001</td>
</tr>
</tbody>
</table>

| Total lease tenures**                      | 21,519 | 110,651,155             | 63.82%         | $5,826,745,681,188 |

#### Agricultural land

<table>
<thead>
<tr>
<th>Tenure Type</th>
<th>Number</th>
<th>Total Area of Land (ha)</th>
<th>% of the State</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Freeholding lease</td>
<td>515</td>
<td>6,391.71</td>
<td>0.00%</td>
<td>468,512,742</td>
</tr>
<tr>
<td>Freeholding lease selection</td>
<td>348</td>
<td>166,858.13</td>
<td>0.10%</td>
<td>37,672,122</td>
</tr>
<tr>
<td>Non-competitive lease converted</td>
<td>7</td>
<td>1.58</td>
<td>0.00%</td>
<td>1,252,502</td>
</tr>
<tr>
<td>Special lease purchase freehold</td>
<td>9</td>
<td>2,342.36</td>
<td>0.00%</td>
<td>3,439,872</td>
</tr>
</tbody>
</table>

| Total freeholding leases**                  | 1430   | 3,082,911.21            | 1.78%          | $693,278,652   |

#### Reserves (Community purpose - internal)

<table>
<thead>
<tr>
<th>Tenure Type</th>
<th>Number</th>
<th>Total Area of Land (ha)</th>
<th>% of the State</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Departmental freehold</td>
<td>190</td>
<td>343.82</td>
<td>0.00%</td>
<td>476,784,133</td>
</tr>
<tr>
<td>Dedicated roads***</td>
<td>3,440,276</td>
<td>1.98%</td>
<td></td>
<td>420,570,202,213</td>
</tr>
<tr>
<td>Protected area estate and state forest</td>
<td>10,270,359</td>
<td>5.92%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other transport (rail) corridors land,</td>
<td>1,060,498</td>
<td>0.61%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>operational reserves and other uses</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

| Total state reserves****                   | 48524  | 16493216.27             | 9.51%          | $571,146,718,432 |

#### Freehold land

<table>
<thead>
<tr>
<th>Tenure Type</th>
<th>Number</th>
<th>Total Area of Land (ha)</th>
<th>% of the State</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Freehold land</td>
<td>2,084,777</td>
<td>43,704,350</td>
<td>25.21%</td>
<td>$4,431,087,502,585</td>
</tr>
</tbody>
</table>

| Total area of Queensland                   | 173,380,000 |

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* Rental value, includes land tenures issued for land below high water mark comprising of about 600 hectares
** Indicated value is instalments owed
*** Of the total road network, 2.1 million hectares are declared stock route.
****Asset value

Source: adapted from DNRM (2013b)

In Queensland the largest proportion of land (65.6 per cent) is held as Crown leasehold land. There are three major types of leasehold tenures:

1. Term leases can be issued for different purposes up to a period of 100 years. They include agriculture, business, communication, education, pastoral production, industrial and infrastructure development and tourism (SDIIC 2013a). In rural leasehold tenures the leases can be granted for a period of 30 years. If certain conditions are met they can be extended up to 75 years (Land Act ss155-155BA).

2. Perpetual leases are held by leaseholder in perpetuity and can be converted to freehold land. To protect family farm and family companies, these leases can be held only by individuals. No more than 2 leases can be held by an individual (Land Act ss145, 146).
Freehold leases are established where a freehold title has been approved, but the leaseholder is paying off the purchase price by annual instalments. The title to the property is not issued until the debt is fully paid (SDIIC 2013a).

Occupational licences and permits (e.g. grazing permits) are issued for a short period of time, and generally cannot be sold or mortgaged (SDIIC 2013a).

In terms of spatial extent, the dominant use (more than 50 per cent of the total area) is ‘rural’ leasehold land, or land leased for grazing (pastoral) and agricultural uses. The average size of a grazing term lease is 20 000 hectares, but leases can exceed 500 000 hectares (SDIIC 2013a). Map 4 shows the distribution of leasehold lands.

Map 4 Queensland leasehold land map
Source: extracted from DNRM (2012)
The Acquisition of Land Act 1967 (Qld) allows government authorities to resume land of any type of tenure, including freehold land and native title, for stated purposes. The compulsory acquisition can occur either with or without the landowner’s agreement. The major purpose of acquisitions is for public infrastructure and development works. However, the Act also incorporates several purposes in relation to the environment which include: protected areas (state land), conservation of koalas in specified areas, soil conservation and works for the protection of the seashore and land adjoining the seashore (Schedule 1).

5.6.2 Indigenous land and native title

Acknowledgement of the rights of Indigenous Australians to own land began in the 1970s after Australia ratified the International Convention on the Elimination of all Forms of Racial Discrimination. The Racial Discrimination Act 1975 (Cth) was adopted by the Commonwealth Parliament to ‘make provision for giving effect to’ the Convention. Among the fundamental freedoms the Convention included the right to own property and the right to inherit it (Article 5(d) (v) and (vi)). Subsequently, the Commonwealth Parliament enacted the Aboriginal Land Rights (Northern Territory) Act 1976 (Cth) providing for land allocation to Aboriginal Australians. Each state and territory has adopted its own laws regulating what land and interests in land can be allocated to Indigenous communities.

In Queensland, allocation of land to Indigenous communities is regulated under the Aboriginal Land Act 1991 (Qld) (ALA) and Torres Strait Islander Land Act 1991 (Qld) (TSILA). The statutes provide for categories of ‘transferrable land’ which are to be granted to Aboriginal communities without a claim (ALA s10) and ‘available State land’ that can be claimed and granted under the provisions of the respective statutes (ALA s24, TSILA s19). In general, claimable land includes unallocated state land, as well as land in national parks (ALA ss24, 30, TSILA ss19, 25). Established freehold and Crown leasehold tenures are explicitly excluded from the scope of available land (ALA s31, TSILA s26). As of 2013, the overall land area transferred to Aboriginal and Torres Strait Islander people in Queensland comprised 3 774 593 hectares, or 2.18 per cent of total area (DNRM 2013a).

Aboriginal and Torres Strait Islander land can be either freehold or Crown leasehold. Most freehold land, however, is practically inalienable because of the restrictions imposed on sale or mortgage of the land (see ALA s100, TSILA s67). A deed of grant or lease also ‘must’ contain a reservation to the Crown of all mineral, petroleum and geothermal resources, and ‘may’ contain a reservation to the state of forest products or quarry material (ALA ss54, 55, TSILA s49, s50). However, where the state receives a royalty under the Mineral Resources Act 1989 (Qld) or the Petroleum and Gas (Production and Safety) Act 2004 (Qld) in relation to Aboriginal or Torres Strait Islander land, the trustee of the land is entitled to receive the prescribed percentage of the total royalty amount. The amount must be used for the benefit of the people for whose benefit the trustee holds the land (ALA s203, TSILA s152).
The resource ownership and land tenure system in Australia became more complex in 1992 with the High Court decision in the case of *Mabo v Queensland (No 2)* 1992 175 CLR 1, which rejected the ‘terra nullius’ doctrine. The High Court recognised a form of ‘native title’, which had survived the property law in Australia and must be treated equally with other titles. ‘Native title’ acknowledges that Indigenous people have rights and interests in the land and resources originating from their traditional laws and customs. While formally called ‘title’, it is not a type of land tenure, as it originates in traditional laws and thus, cannot be granted by the Crown (NNTT 2013).

In response to the Mabo case decision, the Commonwealth Parliament enacted the *Native Title Act 1993* (Cth) (NTA) to provide for the recognition and protection of native title. According to section 223 of the NTA:

(1) The expression “native title” or “native title rights and interests” means the communal, group or individual rights and interests of Aboriginal peoples or Torres Strait Islanders in relation to land or waters, where:

(a) the rights and interests are possessed under the traditional laws acknowledged, and the traditional customs observed, by the Aboriginal peoples or Torres Strait Islanders; and

(b) the Aboriginal peoples or Torres Strait Islanders, by those laws and customs, have a connection with the land or waters; and

(c) the rights and interests are recognised by the common law of Australia.

‘Rights and interests’ in land and resources may include hunting, gathering, or fishing, area access for traditional purposes such as camping and ceremonies, teaching law and custom (NNTT 2013).

The NTA contains extensive provisions with regard to native title. It validates a range of past acts attributable to the Commonwealth and state and territory governments, which took place before 1 January 1994, and intermediate acts, which took place before 23 December 1996, and that have extinguished the existence of native title. In general, granted freehold estates and leases (i.e. perpetual leases, freehold leases), and construction or establishment of public works, completely extinguish native title. Non-exclusive possession acts which involve non-exclusive agricultural leases, or non-exclusive pastoral leases, extinguish native title to the extent of any inconsistency (NTA s23A). The NTA specifies detailed requirements for the payment of compensation, if certain acts extinguish or impair native title. Indigenous land use agreements are important instruments for dealing with future interests in land impacting upon the native title (NTA, Division 3, Subdivisions B-D). The Federal Court of Australia is responsible for the determination of native title and for decisions regarding compensation for the loss or impairment of native title.

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28 Note: the extent of traditional rights was further specified by the High Court in 1996 in *Wik Peoples v Queensland* (1996) 187 CLR 1. In the Wik case the High Court proclaimed that leasehold tenures do not extinguish native title. However, pastoral leases issued prior to 1 January 1994 were recognised as valid grants allowing the rights of pastoralists to prevail to the extent of any inconsistency.
In Queensland, the *Native Title (Queensland) Act 1993* (Qld) (NTA Queensland)\(^{29}\) complements the regulation and validates past acts of the Queensland Government that may have affected or extinguished native title. Among others, the NTA Queensland confirms the state ownership of all natural resources and the right to use, regulate and control the flow of water and all existing fishing access rights (s17).

### 5.6.3 Mining tenements

Mining tenements are important types of tenure affecting the use of land in Australia. In Queensland several laws (further called Resource Acts) regulate access and extraction of mineral, petroleum and geothermal energy resources: *Mineral Resources Act 1989* (Qld) (MRA), *Petroleum and Gas (Production and Safety) Act 2004* (Qld) and *Geothermal Energy Act 2010* (Qld). With limited exceptions the Resource Acts vest ownership of minerals, petroleum products and geothermal energy in the Crown. The Queensland Government on behalf of the Crown can grant this interest irrespective of established interest in land. The rights can be granted over private (freehold or leasehold) land without the owner’s consent. However they are subject to compensation for loss of the use of the land (e.g. MRA s279). An agreement must also be reached with native title holders (e.g. MRA Schedule 1A).

The Resource Acts provide a framework for granting authorities to explore and extract the resources. Broadly, all mining tenements can be divided into two major groups: (1) tenements that permit the resource company to explore the land over which the tenement is granted (prospecting permits, exploration permits, mining claims), and (2) tenements that permit extracting respective resources (mining/petroleum/geothermal leases). The environmental impacts of resource extraction activities are regulated separately under the EPA (see section 5.5.2.2). The Queensland government is entitled to the royalties from extracted resources calculated in accordance with provisions of respective Resource Act (e.g. MRA Chapter 11). Mining royalties form a significant part of government revenues (see section 5.4.2).

Mining and petroleum extraction activities are exempt development under the SPA and are regulated separately. State interests with regard to extractive resources are formulated in the *State Planning Policy* (DSDIP 2013c) aiming to limit encroachment of sensitive development in resource areas. In practice, subject to imposed conditions and few other restrictions established by the legislation (e.g. ‘reserve land’ as defined in MRA), resource extraction can occur in any area of the state having resource reserves. Mining tenures cannot be granted over state protected areas (see below), some parts of wild river areas declared under the *Wild Rivers Act 2005* (Qld), and areas identified as high value agricultural land under the provisions of the *Strategic Cropping Land Act 2011* (Qld).

\(^{29}\) Note: according to the Australian Constitution, the Commonwealth Parliament has jurisdiction over the matters relating to native title. In case of inconsistency the Commonwealth legislation prevails.
5.6.4 Land tenure and biodiversity conservation

Protected area land holdings and state forests is the largest land estate which contributes to biodiversity protection. In Queensland, all national parks are established on state land set aside for nature conservation. The Nature Conservation Act 1992 (Qld) (NCA) provides several classes of protected areas, each having a different management regime and permissible uses of land. The NCA also includes provisions which enable granting national parks to Indigenous communities on conditions involving a perpetual leaseback to the state. National parks (Aboriginal Land) and National Parks (Torres Strait Islander Land) must be managed in accordance with the principles established for a national park, and in a way, as far as practicable, that is consistent with any Islander or Aboriginal custom and tradition (NCA ss19, 20).

The NCA explicitly prohibits granting mining interest, geothermal tenure, or greenhouse gas storage authority in relation to all classes of national and general regional parks (s27). Mining interest can be granted over state land in regional parks which are declared as a ‘resource use area’. Stock grazing in national parks is not allowed, but permits have been recently issued in several protected areas for emergency drought relief (NCA s173S, see section 5.5.2). State forests designated for timber extraction are declared and managed under the Forestry Act 1959 (Qld). Stock grazing permits, resources permits authorising the removal of quarry material, as well as fossicking licences can be granted in state forests (ss35, 46, 46A). State forest areas are not excluded from granting of mining interest.

Conservation on private land usually limits the ability of the landholder to use the land for agricultural production or other intensive uses, which, in turn, affects its market value. In general, the statutes incorporate restrictions on the regulatory authorities to change the established use of land, affecting its use or diminishing its value, without compensation (e.g. CPMA s150, SPA s704). Therefore, of particular importance are voluntary agreements and incentive mechanisms. In Queensland, several regulatory instruments have been developed to increase the private land area allocated for biodiversity conservation.

The NCA provides for two categories of private protected areas: nature refuge and coordinated conservation areas. A nature refuge is established to ‘conserve the area’s significant cultural and natural resources’ (s22(a)). Coordinated conservation areas are established in cases where the protected area is managed by several landholders (s23). Conservation areas on private land (freehold and Crown leasehold) are established based on voluntary agreements negotiated individually between the landholder and the Queensland government. The agreement allows for controlled use of an area’s resources and takes into account landholder interests (s22). The landholder retains the ownership/leasehold rights. Conservation agreements, recorded by the

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Prohibition does not apply to a survey licence and a pipeline licence for oil and gas supply activities, as well as existing uses (s27(2))
Registrar under the provisions of the NCA, are binding on the successors in title to the landholder, and persons who have interest in land (ss51, 134).

Conservation covenants can be also established under the provisions of the *Land Title Act 1994* (Qld) (LTA) and the *Land Act 1994* (Qld) (Land Act). Both Acts allow covenants to be registered in favour of the state, another entity representing the state, or a local government for the purposes specified in the legislation. They allow registering a covenant, which is aimed at preserving a native animal or plant or natural or physical feature of the lot, which has cultural or scientific significance (LTA s97A(3)(b), Land Act 373A(4)(b)). If the land is leased from the state, the Minister administering the Land Act must issue consent to a covenant affecting that land. A covenant may be registered even if the ‘covenantor’ and the ‘covenantee’ are the same entity (LTA s97A(2A), Land Act s 373A(2A)).

Registration of covenants or conservation agreements is a voluntary measure, and they are supported by several incentive mechanisms. The Land Act includes provisions for extended leases (50 or 75 years) of large areas of rural leasehold land (1000ha and more), which contain areas of conservation value, provided the land is in good condition and the leaseholder has entered into a conservation agreement or covenant (see section 5.5.2 regarding recent policy change). Common mechanisms are stewardship payments, or financial assistance provided under various Queensland (e.g. NatureAssist) and national (e.g. Caring for Country) programs, as well as though private funds. Since 2000 land owners who have entered into conservation covenants under national environmental programs can be eligible for an income tax deduction, or concessional capital gains tax treatment. The owner, who does not receive any material benefit, can claim as a deduction the difference between the market value of the land and its decreased market value (ATO 2012).

Conservation on private land, however, has one major limitation - nature refuges and coordinated conservation areas, covenants and other conservation agreements, do not prohibit the Queensland government from issuing mining exploration permits and leases. According to Adams and Moon (2013), in 2011 there were 273 current mineral exploration permits within the boundaries of 149 of the 379 gazetted nature refuges, and 186 of them were approved after the nature refuge status was assigned to the area.

Queensland local governments are also authorised to apply incentive measures and collect resources for biodiversity conservation. Section 120 (1)(e) of the *Local Government Regulation 2012* (Qld) allows granting rates concession, which ‘will encourage land that is of cultural, environmental, historic, heritage or scientific significance to the local government area to be preserved, restored or maintained’. Local governments are also authorised to collect levies to support purchase of ecosystem assets. As of 2011, 57 non-indigenous and 2 indigenous local

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31 Note: 14 indigenous councils were not included in the dataset
governments were responsible for the management of 57 201 hectares of parks and gardens (DLGCRR 2013a).

This study could not obtain precise data summarising the extent of rates concessions applied by Queensland local governments to support biodiversity conservation. According to the reported financial data (DLGCRR 2013a), in 2010-2011 environmental levies have been applied by 20 councils in total generating $103.8 million. Half of this amount ($52.8 million), however, was collected by the Brisbane City Council, with the other major contributor being Gold Coast City Council ($13.58 million).

5.7 Institutional diagnostics: barriers and bridges to biodiversity conservation

Human pressures on the natural environment have led to increasing loss of biodiversity in Australia and in Queensland. It is acknowledged that this problem requires an adequate institutional response. As stated in the national report by the Biodiversity Decline Working Group (BDWG 2005:ii):

Arresting the decline in biodiversity will require a range of institutional changes to provide adequate planning and management frameworks and integration of effective delivery mechanisms.

Biodiversity protection is an ongoing process and there are no fixed measurable endpoint goals. An overarching need, as stated in the national and state level strategies, is ‘to halt’ biodiversity loss (NRMMC 2010, DERM 2011a).

Most of Australia’s land resources and their uses are already ‘pre-planned’ and many of the current environmental problems are the legacy of past decisions. More than 90 per cent of Queensland’s land is allocated for development activities, ranging from intensive uses such as urban, industrial development and mining, up to extensive uses such as livestock grazing (see section 5.3.2). In this context, institutional effectiveness in protecting biodiversity can be primarily linked to measures that:

(1) contribute to shifts in land uses along ‘intensive use → extensive use → protected areas’ chain (e.g. state forests → forest reserves → national parks);

(2) place constraints on the land use activities adversely affecting the natural environment (e.g. limitations on vegetation clearing); or

(3) require/promote rehabilitation of affected areas.

This section discusses the major findings from the analysis of the broader network of rules constituting the Queensland Planning System. The findings are discussed using the structure of the diagnostic framework (Table 6 in chapter 4) and guided by several groups of questions adapted from the framework.
5.7.1 Regulatory scope and outcomes

To what extent does the allocated regulatory scope require or enable regulatory authorities to:
- consider biodiversity protection outcomes;
- prioritise protection of the natural environment (biodiversity) over competing or conflicting land uses;
- cover the scope of interventions that produce adverse effects (i.e. leaves no gaps);
- respond to observed or predicted changes in the biophysical environment or changes in the environmental knowledge or values?

The overall capacity of regulatory authorities to achieve protection of the natural environment is determined by the allocated regulatory scope. In statutory systems the scope is determined by the overarching purpose and/or objective statements and geographical, functional (i.e. regulated interventions), ecosystem property (e.g. vegetation) and other boundaries. In strategic frameworks the authorities are largely guided by established strategic and operational outcomes and targets.

The commitment to ‘ecologically sustainable development’, which includes biodiversity protection and related (protection of habitat, environment, ecological processes) outcomes, is incorporated in many statutory frameworks. It is set out in the federal EPBC Act and in the majority of the Queensland statutes regulating the use of land and extractive environmental resources (McGrath 2010). However, below this level, the regulatory approaches and allocated priorities diverge significantly.

In Queensland, biodiversity (nature) protection receives priority only in specifically designated areas such as national parks. Outside these areas, the statutes do not prescribe clear criteria how or under what conditions biodiversity protection values should be prioritised over competing or conflicting interests. Furthermore, each statute covers only a fraction of the resource distribution authorities (e.g. fish, water, forest resources) or pressures affecting environmental values (e.g. vegetation clearing, pollution, infrastructure construction). As a result, specific goals and objectives pursued under one regulatory framework may not be taken into account in instruments designed or applied under other frameworks. This is particularly evident in case of gazetted nature refuges which remain vulnerable to ‘mining invasion’ (Adams & Moon 2013).

It can be argued that under the established tenure system (see sections 5.6.3 and 5.6.4), priority in Queensland has been given to mining, petroleum and geothermal energy extraction and related infrastructure, followed by agriculture and urban development.

This chapter does not include a detailed analysis of all potential threats (e.g. environmentally relevant activities) and their regulation. Even so, one significant regulatory gap can be identified in the protection of ecosystems from grazing pressures. Apart from the Great Barrier Reef catchments regulated under the EPA, grazing pressures, including those impacting upon riparian
and riverine vegetation, remain largely unregulated. Vegetation destruction by stock is explicitly excluded from the ‘clearing’ definition under the VMA (see chapter 6 for details). This gap is also not covered by the EPBC Act which has little direct impact on agricultural activities (Macintosh 2012). It can be predicted that this gap will widen with the recent changes in the Queensland government’s policy on the control of the rural leasehold land condition and management practices (see section 5.5.2.2).

Apart from specific cases, such as breach of statutory provisions or imposed conditions, environmental restoration is not covered by statutory frameworks. The legislation generally seeks to maintain the status quo or minimise further pressures. This can be explained by the accepted ‘good legislation’ principles which preclude the regulators from imposing adverse retrospective obligations on land users (see section 5.4.3). While exceptions made in the public interest are possible, such measures are highly unpopular and some form of compensation is commonly expected from the government (see Kehoe 2009 for the VMA history). Therefore, effectiveness and scope of restoration activities largely depends on non-statutory measures, such as community-based NRM, Landcare and other programs. Statutory frameworks can mainly assist with securing agreements (e.g. application of covenants) and their enforcement.

Both statutory and strategic frameworks are dynamic. Acts of Parliament allocate considerable degree of flexibility to regulatory authorities with regard to the design and application of authorised instruments. As indicated in the state of environment reports (SOEC 2011, DEHP 2012), many regulatory changes have occurred in response to environmental problems. The move towards increasing levels of environmental protection, however, is constrained by the vested interests of market actors requiring security for investments (e.g. clarity of clearing restrictions, stability of resource allocation authorities). It also depends on the level of resources available to the regulator to compensate for the loss in land value or align incentives (see section 5.7.6 below).

### 5.7.2 Distribution of roles and responsibilities

| To what extent does the institutional system enable the allocation of regulatory authority to participants who: |
| - have commitment (e.g. institutional role) to protect environmental values; |
| - have capacity (e.g. human resources, finances) to undertake allocated functions; |
| - hold relevant environmental information/knowledge or have the capacity to obtain required information/knowledge? |

Environmental regulation largely depends on the commitment and capacities of the regulatory body to perform allocated functions. As noted by Sabatier and Mazmanian (1980:547):

No matter how well a statute structures the formal decision process the attainment of statutory objectives which seek to significantly modify target group behaviour is unlikely unless officials in the implementing agencies are strongly committed to the achievement of those objectives.
Initial distribution of regulatory responsibilities, as well as revenue raising capacities between federal and state/territory governments has been determined by the Australian Constitution. Regulatory roles in environmental matters have been further specified in the IGAE allocating the federal government responsibility for ‘matters of national environmental significance’ which includes the obligations under the Biodiversity Convention. The Commonwealth has both the powers to override state/territory legislation in many environmental matters (McGrath 2010) and the right to collect and distribute the largest part of taxation revenues (see section 5.4.1). However, in the implementation of its policies the federal government depends on the states and territories for ‘on ground’ human resources and implementation control. Consequently, since the 1990s the major role taken by the federal government in biodiversity protection and other environmental matters is that of initiator, coordinator and co-funder of national environmental policies (see section 5.5.1).

Over the years, the federal government has initiated many regulatory reforms in the environmental field and contributed to the development of an extensive knowledge base. Most recently, however, it demonstrates decreasing political commitment to interfere in environmental regulation. As indicated by the recent changes in the COAG structure (see section 5.5.1), the ‘environment’ ceased to be a priority matter for intergovernmental coordination, at least for the next 12-18 months. Similarly, the recent decision to delegate development approval powers under the EPBC Act to state governments is seen by many environmental lobbies (e.g. ANEDO 2013) as a major threat to protected environmental values.

The state/territory governments have the primary regulatory power with regard to the allocation of land and environmental resources and biodiversity protection. In general, all Australian jurisdictions have made progress in establishing directions for biodiversity protection. Continuous increase in the extent of protected areas is a national trend noted in many environmental reports (OECD 2008, SOEC 2011). In Queensland, the total terrestrial area allocated for biodiversity conservation has increased from 2.1 per cent in 1990, up to 5.1 per cent in 2012 (DEHP 2012). Introduction of the VMA framework and gradual expansion of its scope has significantly decreased the rate of vegetation clearing outside protected areas.

Similarly, the commitment of the Queensland government to protect environmental values is decreasing. As the established strategic direction indicates, in the nearest future an expansion of regulatory measures directed towards biodiversity protection will be put ‘on hold’. Moreover, some of the established protection regimes will be reversed, or their levels diminished to enable expansion of agriculture and extractive industries (see section 5.5.6).

Organisational structures of all levels of government are dynamic and largely driven by the election cycle. As a result, discussing the effectiveness of distribution of the environmental portfolios among the departments or establishing some long term trajectories is difficult. There is no capacity assessment data available to discuss ‘portfolio scope – capability’ match in the
Queensland government. However, in light of the current strategic direction (see section 5.5.6), power and resource distribution within the government raises several concerns. They have been expressed in the recent Report of the Queensland Audit Office (QAO) tabled in the Parliament in April 2014 (QAO 2014) (see Box 1).

The current distribution of the regulatory portfolios among the departments in Queensland government (see Table 10 in section 5.5.2) is also of concern. Departments have different objectives and priorities and, consequently, different values can be attached to environmental outcomes. Distribution of ‘environmental’ portfolios to departments pursuing development goals may lead to neglect of environmental regulation. Currently, significant discrepancies can be found between the overarching ‘strategic’ visions, goals and performance indicators of the departments and the statutory goals they are expected to achieve as environmental regulator. For example, no explicit commitment to protect remnant vegetation, or to consider environment as a user of water resources, can be found in annual reports of the DNRM responsible for the administration of the VMA and the Water Act 2000 (Qld) (see DNRM 2013a).

Box 1 Queensland Government: environmental performance audit

**Background:**
In 2013, the Queensland Audit Office (QAO) undertook a performance audit of two departments of the Queensland Government:
- the Department of Environment and Heritage Protection (DEHP) which administers the Environmental Protection Act 1994 and is the environmental regulator of resource and waste industries;
- the Department of Natural Resources and Mines (DNRM) which administers the Mineral Resources Act 1989 and the Petroleum and Gas (Production and Safety) Act 2004 and is the regulator of the resources industry under these Acts.

**The objective:** to determine whether the supervision, monitoring and enforcement of environmental conditions for resource and waste management activities is effective and protects the state from liability for rehabilitation and the environment from unnecessary harm.

**QAO conclusions:**
EHP is not fully effective in its supervision, monitoring and enforcement of environmental conditions and is exposing the state to liability and the environment to harm unnecessarily.
Poor data and inadequate systems continue to hinder EHP's planning and risk assessments. As a result, EHP cannot target its monitoring and enforcement efforts to where they are most needed. This situation is exacerbated by the lack of coordination and sharing of relevant information across agencies, particularly between EHP and NRM.
Under its regulatory strategy, EHP focuses less on regulating access to the market—through the assessment and approval of applications for environmental authorities and their conditions—and directs its resources to increased effort in enforcement. To be effective, this approach requires high quality, relevant, reliable and timely data.
Poor data have hampered past approaches to effective environmental regulation of the mining and waste industries. This issue is now brought into even sharper relief under the current regulatory strategy, based as it is on government policy to ease the burden on industry caused by regulation and its associated bureaucracy—red tape and green tape specific to environmental issues.
Green tape reduction aims to reduce costs for industry and government while maintaining environmental standards. To be effective, this requires the appropriate allocation of resources and effort according to the risks involved and outcomes to be achieved. The inability of both departments to assess risk effectively, and to target and coordinate their resources appropriately reduces confidence for the community that the environment is protected and standards have been met.

Source: information extracted from QAO (2014)
Australian local governments play a minor regulatory role in environmental (biodiversity protection) matters. Regulatory functions are delegated by the state/territory governments which largely determine the overall policy direction. Local governments in Queensland differ significantly in their administered areas and population, and their capacities and interest to undertake environmental regulation cannot be generalised.

5.7.3 Choice of policy instruments

<table>
<thead>
<tr>
<th>To what extent does the institutional system enable environmental regulators to develop, apply or change regulatory instruments which:</th>
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<tr>
<td>- confer protective status on ecosystems or their properties;</td>
</tr>
<tr>
<td>- limit or prohibit actions which might adversely affect ecosystems or their properties;</td>
</tr>
<tr>
<td>- provide for incentive measures to protect ecosystems and biodiversity;</td>
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<tr>
<td>- incorporate monitoring procedures, enforcement and remedies?</td>
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Overarching regulatory frameworks such as Acts of Parliament and ‘whole of government’ strategies set out the overall purpose and scope of regulation. Achievement of intended outcomes largely depends on the set of instruments (e.g. incentive/disincentive mechanisms, plans, regulatory maps, policies, sanctions), which the responsible agency is authorised to develop or apply. The effectiveness of such measures can be determined by the extent to which these instruments protect environmental values and/or reduce pressures.

At the national level, matters of national environmental significance are protected through the definition of their scope (e.g. listing of threatened species), and the approval system for actions having significant impact on these matters. These instruments, however, are limited in terms of the extent to which they can provide an ‘on ground’ protection of biodiversity. As Macintosh (2012) concludes, the regulatory regime of the federal EPBC Act had almost no impact on deforestation. He establishes that between July 2000 and July 2008 the approval system was applied to a total of 10 agricultural land clearing projects, involving removal of 6200 ha of vegetation, which is less than 0.2 per cent of total national deforestation over the period.

Of significant importance at the national level is an extensive policy system which incorporates a number of strategies covering such matters as greenhouse gases, biodiversity, pollution and water management (see section 5.5.1.2). The ‘steering power’ of the federal government lies in funding mechanisms designed to support their implementation (e.g. National Water Initiative, Forest Policy). At the same time, the effectiveness of these instruments is hard to evaluate, as outcomes vary. For example, the National Biodiversity Strategy 1996 failed to achieve its intended outcomes, which has been partially attributed to the lack of commitment by state and territory governments (Australian Government 2009, NRMMC 2010).

The taxation power provides the federal government with another powerful set of tools which can directly influence the interest of landholders to protect native ecosystems. Taxation incentives have been applied to support extension of private protected areas (see section 5.6.4).
The carbon trading scheme and biodiversity market systems, however, have developed slowly. The design of a carbon trading scheme, and its influence on business environment, has been subject to fierce political debate between the two leading parties of the Federal Parliament promoting different cost distribution approaches.

The Queensland government has introduced a wide scope of regulatory instruments to address environmental problems. They include: declaration of terrestrial and marine conservation areas, protected plants and habitats, reduction of adverse environmental impacts through licensing controls, as well as provision of incentives to promote sustainable land management practices (DEHP 2012). Moreover, the land tenure system allows the government as land owner to control the clearing of native woody vegetation on almost 70 per cent of terrestrial area (see Table 12 in section 5.6.1).

Partially due to reporting requirements (see section 5.7.4 below), there is limited data on the operation and effectiveness of particular instruments. The significance of some preventive measures, such as declaration of wild river areas and licensing of environmentally relevant activities, is hard to determine as they need to be examined in the pressure context. For example, it is relatively easy to establish protected areas on land unsuited for agricultural or other uses (see e.g. Map 2). However, policies that restrict the use of agricultural land, or land with high market value, are hard to introduce and may require years of negotiation.

In light of development pressures, the most successful regulatory system in Queensland that has led to measurable ‘on ground’ results, has been the land clearing control under the VMA. Despite the political controversy, the introduction of the VMA system, and gradual expansion of its scope, has led to significant improvement in the retention of vegetation cover. The 2011 Queensland state of environment report (DEHP 2012) (see Figure 17) and the study by Macintosh (2012) identify a direct correlation between the introduction of the VMA and reduction in the deforestation rate.

![Figure 17](image_url) Average annual clearing rates for remnant vegetation from 1997 to 2009
Source: extracted from DEHP (2012:26)
Despite their success, the VMA and instruments developed under other regulatory frameworks, do not preclude further pressures and vegetation clearing continues. Significant damage has been done to the regulatory system by the LNP government’s weakening of the protective ‘strength’ of many instruments, such as wild river codes and declarations and restrictions on undesirable activities in national parks (see section 5.5.2.2). Pollution risk has increased as a result of changes made in the approval system of environmentally relevant activities (see QAO conclusion in Box 1). Changes in the VMA and their environmental impacts will be examined in more detail in chapter 6.

Queensland local governments are authorised to apply regulatory measures to control vegetation clearing in local government areas and to establish protection regimes. Rate concessions and environmental levies are also among available instruments. However, as the data indicates, these tools are applied only in limited cases, and their extent cannot cover regulatory gaps. The limited interest of local governments to undertake their regulatory roles can be partially explained by current funding arrangements (see section 5.7.6 below).

5.7.4 Decision-making integration

<table>
<thead>
<tr>
<th>To what extent does the institutional system enable cross-jurisdictional coordination and cooperation in environmental regulation?</th>
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<tbody>
<tr>
<td>To what extent does it allocate the control over joint decisions to the actors who:</td>
</tr>
<tr>
<td>- have a commitment or responsibility (e.g. institutional role) to protect environmental values;</td>
</tr>
<tr>
<td>- represent the community benefiting from the services of affected ecological systems, including adjacent and otherwise connected systems?</td>
</tr>
</tbody>
</table>

Jurisdictional fragmentation is an inevitable feature of modern environmental governance. Due to the variety and scale of pressures, any single regulator is usually deficient in the authority to achieve outcomes leading to the ultimate protection of environmental resources and biodiversity. Long term protection of environmental values, therefore, is dependent upon the integration of environmental considerations across regulatory portfolios. There are several governance techniques available (see Ross & Dovers 2006 for a review). However, their application and outcomes will largely depend on the interests represented by involved regulators and their decision-making power.

At the national level, the need for intergovernmental cooperation and coordination is already embedded in the power distribution set out under the Australian Constitution. The relations between the three tiers of government are not defined by a strict hierarchy. At the national level, intergovernmental coordination is largely organised through administrative structures such as the COAG and supporting ministerial councils (see section 5.5.1). The national policy-making process is based on negotiations and, consequently, is influenced by different ‘informal’ power mechanisms, which cannot been examined within the scope of this study. However, the common feature of adopted national policies and strategies, including environmental, is that the states and territories remain the final decision-makers with regard to their implementation.
In Queensland, there is a large number of statutory and non-statutory frameworks providing for the regulation of environmental systems and threatening processes. The organisational structure is dynamic and administrative units and their portfolios are shifted (at least) on a 3 year election cycle basis. While there are no benchmarks for an ‘environmentally’ optimal organisational structure, fragmentation of information systems and the need to re-establish coordination processes is an ongoing problem (see QAO conclusions in Box 1). On the one hand, a significant effort of the Queensland government has been put into streamlining and simplifying legislation and developing an integrated decision-making system such as the SPA. On the other, the policy integration process, in particular since 2012, is driven by two major agendas: cost saving and the ‘four pillar’ economy growth. As a result, several integration measures have the potential to diminish consideration of environmental interests.

On the broader scale, government departments do not operate in isolation. Development and approval of regulatory instruments is a complex process with many internal quality control and ‘veto’ points (see Queensland Legislation Handbook (DPC 2014)). Even decision-making control that is allocated for ‘environmental’ department protection measures will not be adopted, if they contradict an overall policy direction taken by the government (see Figure 16 above). Therefore, regulatory reforms and approaches need to be interpreted in the context of direction set by the political party in government (see discussion in section 5.7.6).

5.7.5 Environmental information

| To what extent does the institutional system enable the environmental regulators: |
|-----------------|--------------------------------------------------|
| - to capture decision-making situation and evaluate potential environmental outcomes and/or impacts produced by particular regulatory solutions; |
| - to obtain information on past, existing or potential impacts (pressures) of regulated interventions on ecological systems or their properties; |
| - reduce uncertainty and gain shared understanding about the environmental problem under consideration? |

There are many gaps in knowledge about Australia’s native ecosystems and their functional processes. As major national reports identify (Australian Government 2009, SOEC 2011), data gaps create significant problems for evidence-based policy-making in biodiversity protection and related matters. A lack of knowledge about the status of biodiversity and appropriate responses can become a driver of biodiversity loss (BDWG 2005).

There are several separate research bodies established and financed by both national and Queensland governments that carry out and coordinate environmental research (CSIRO, BOM, Queensland Herbarium). Government departments holding ‘environmental’ portfolios (e.g. DNRM in Queensland) are responsible for the development and maintenance of environmental data bases required for the performance of regulatory functions, but there are no statutory prescribed processes for scientific information to be delivered to, and incorporated in, the regulatory system. At the same time, there are few, if any, environmental regulatory instruments
(e.g. offset policies, carbon trading scheme, distribution of fisheries or water resources) which would not be based on scientific knowledge.

In biodiversity protection, IBRA bioregional classification provides the biological foundation for selection of representative areas for the National Reserve System (Australian Government 2009). It also forms the basis for the regional ecosystem framework established under the VMA for vegetation clearing control in Queensland. The Queensland government applies satellite monitoring and spatial mapping (SLATS) program for remnant vegetation mapping and monitoring. As stated in the Queensland Biodiversity Strategy (DERM 2011:48):

> The current regional ecosystem mapping data across two areas of remnant and pre-clearing provide an exceptional basis on which to build understanding of biodiversity health in Queensland. Regional vegetation mapping uses satellite imagery, in combination with recent aerial photography and field-based inspections to confirm data accuracy. The Biodiversity Assessment and Mapping Methodology builds on these sound foundations to provide a consistent approach for assessing biodiversity values at the landscape scale in Queensland.

Several performance reporting processes have been established for regulatory systems. Both federal and Queensland governments publish comprehensive state of environment reports covering several environmental themes, such as the state of the coastal zone, availability and quality of water resources, biodiversity, vegetation cover and soil condition. Such reporting is prescribed by the legislation (in Queensland under the EPA and the CPMA). However, as examined by McGrath (2010), and observed in the data gathering process for this study, they contain limited evaluation of regulatory responses. Most of them describe selective response measures, and largely remain silent on regulatory gaps and implementation problems. More detailed performance audits occur on an ad hoc basis (see Box 1), or precede major reviews of legislation or strategic frameworks.

In Queensland, strategic environmental assessments are not required under the legislation. The established policy assessment practice is that of regulation impact analysis (RIS), which is mainly aimed at reducing regulatory burdens. Adopted RIS guidelines (Queensland Government 2013e) require assessment of a variety of regulatory impacts, including environmental. The focus, however, is on environmental economic valuation and evaluation of costs and benefits to the community.

An integral part of the government’s accountability system is an annual report prepared by each department. However, as observed in the review of 2012-2013 annual reports of departments administering ‘environmental’ portfolios (see sources under Table 10), little information is available on the performance of regulatory functions and their outcomes. Under the current system all agencies are required to report on their financial performance and the delivery of ‘election commitments’ of the government. In other words, systematic evaluation of environmental effects of regulatory measures is lacking.

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32 Note: the only comprehensive strategic assessment in Queensland is being conducted in accordance with the EPBC Act for the Great Barrier Reef World Heritage Area and adjacent coastal zone.
5.7.6 Incentives and disincentives

<table>
<thead>
<tr>
<th>Incentives and disincentives</th>
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<tbody>
<tr>
<td>- economic valuation of biodiversity or ecosystem services;</td>
</tr>
<tr>
<td>- incentives for government actors to uptake the position of an environmental regulator;</td>
</tr>
<tr>
<td>- incentives for environmental regulators to adopt measures that limit/constrain activities placing pressures on ecological systems and biodiversity?</td>
</tr>
</tbody>
</table>

The effectiveness of legislation, or any other institutional arrangement, depends on the commitment of the regulator to develop appropriate instruments, monitor their implementation, and support the achievement of intended outcomes by incentive or enforcement measures. There is a broad range of institutional (e.g. economy, markets, international agreements) and non-institutional (e.g. leadership, media, extreme events) factors that can influence environmental policy direction. While their causal role is difficult to isolate, several institutional arrangements can be identified as the drivers of environmental policy outcomes in Queensland.

Economic growth measures such as the GDP and GSP are important measures of a government’s performance. Australian states and territories, in particular Queensland rely on a resource based economy, depending on such outputs as mining, crop agriculture and beef production providing significant contributions to GSP growth. The Queensland government’s strategy to boost economic growth, building on long established ‘resources and agriculture’ pillars inevitably leads to ‘development-environment’ conflict. Biodiversity protection measures involve trading land resources required to ‘boost’ (i.e. double production by 2040) agriculture and forestry sectors against the use of land with far less potential to support economic growth. To this end GSP, which does not incorporate environmental transactions (DEHP 2012), can operate as a ‘perverse’ performance indicator for governments focusing on short term political gains.

Another set of rules creating disincentives for environmental protection measures is the taxation system and strategy. The Queensland government’s own revenues are mostly derived from payroll tax, gambling, mining royalties and property transfers (see Figure 14 above). The taxation strategy is based on the maintenance of a low tax base to create an attractive environment for investment. A lower than national average tax base is also seen as an advantage in interstate competition (Queensland Government 2013b). Consequently, increasing tax or introducing new taxes or levies, including environmental ones, are highly unpopular measures. In the light of a growing fiscal deficit in the budget (see section 5.4.2) little incentives exist to support investments in environmental sciences and data acquisition, to expand protected area estate, or to compensate landholders for conservation efforts. Similarly, limiting mining impacts in nature refuge or wild river areas, or placing high offset requirements for businesses, implies a reduction in potential revenue streams.

The economic valuation of biodiversity as a public good and service provider is still in an early stage of evolution. Carbon offset markets are recent policy experiments and their evolution and
effectiveness remains to be seen. Biodiversity protection on the state reserve lands, including forest reserves does not generate significant income. Under a utilitarian approach, which dominates the current political environment in Queensland (see Table 11), protected areas are defined as ‘recreational’ service providers, required to boost tourism and ecotourism industry. Such an approach implies the prioritisation of coastal protected areas, which are the major tourist attractors. Therefore, with declining interest of the federal government in environmental matters, as indicated by the reforms in the COAG structure (see section 5.5.1), inland national parks and other areas relevant for conservation of other types of ecosystems may be at risk of becoming an unwanted ‘expense position’, and degraded to state forests or allocated for cattle grazing (e.g. ‘emergency drought’ measures).

More fundamentally, many pro-development changes in the Queensland government’s environmental policies may be linked to the environmental preferences, values and concerns of the electorate. Current environmental regulations and strategies are not the result of ‘enlightened’ recognition of the right of species to exist, or an appreciation of the uniqueness of the native environment. They have largely evolved in response to widespread degradation of land and water resources, threatening the viability of rural economies (e.g. NHT fund) and coastal settlements. To this end, the VMA framework can be regarded as a ‘crisis regulation’ to stop the environmental degradation process (Kehoe 2009).

Environmental protection forms part of the political agenda. Calls to ‘protect koalas, dugongs and turtles’, which are important determinants of the Queensland government’s policy priorities and investments (see section 5.5.2), are far from being a rational response to the problems identified in the state of environment reports. It is evident, that despite their wide recognition, ESD principles are not regarded as ‘fundamental’ as the need for economic growth.

Australian local governments are frequently seen as important actors in environmental governance. However, local governments neither distribute extractive environmental resources nor are entitled (with some exceptions) to collect fees or royalties (see section 5.4.2 for revenue sources). The income structure which builds on local taxation measures contains little incentives for local governments to undertake the role of environmental regulator. Furthermore, many local governments are under continuous pressure to generate revenue streams which would address the needs of local communities (see Productivity Commission 2008). Therefore, it is hard to expect that they would use their scarce financial resource base for biodiversity conservation measures.

5.8 Summary and conclusions

This chapter has been guided by an overarching question regarding the effectiveness of the Queensland Planning System in achieving biodiversity protection outcomes. Based on documentary sources it examined a broad network of rules determining the land use and
environmental resource use pattern in Queensland, and its effects on the state of natural systems and biodiversity. Strengths and weaknesses of the established ‘rules of the game’, and potential effects on biodiversity protection outcomes were discussed using the diagnostic framework developed in the first part of this study.

The uses of Australian land and environmental resources have been determined by over 200 years of European settlement. Australia’s and Queensland’s relatively small but increasing population is mainly concentrated in the coastal zone. Extensive areas of land are allocated for agricultural use. Loss of native vegetation and ecosystem function is a continuing problem requiring changes in institutional systems. Targets to arrest biodiversity and habitat loss form part of Australia’s international obligations.

Australia’s governance is a complex system, consisting of various statutory and non-statutory frameworks interacting across the levels of governance hierarchy. Environmental regulation is embedded in established system of regulatory roles and responsibilities of the three tiers of government. The system is dynamic and, along with pressures, the scope of regulated activities continues to expand. However, a significant gap still remains between restrictions required to address the problem of biodiversity loss and those placed by the regulators on resource use activities.

Over the last two decades, the state of Queensland has introduced various regulatory instruments to protect native ecosystems and limit development-related pressures. Significant progress has been made in expanding the protected area network, as well as in the introduction of vegetation clearing restrictions. However, regulatory gaps still remain in addressing pressures resulting from many agricultural activities, in particular pastoral production. Environmental regulation is also impeded by a lack of detailed information on environmental impacts of adopted regulatory measures. Above all, there is a problem of economic and budgetary incentives that drive policy decisions in favour of agricultural and resource extraction development.

The gradual improvement of environmental regulation in Queensland has been interrupted by recent changes in the policy direction taken by the LNP government since 2012. Reforms of the regulatory framework were driven by development considerations. The ‘four pillar’ economy growth strategy has been supported by planned increase in agricultural production, which already exploits 80 per cent of the state’s land resources. Consequently, many previously established protective mechanisms have been either removed or wound back so as not to impede development.

Introduced changes and the new strategic direction taken by the Queensland government suggest that in the near future biodiversity decline will be accelerated.
Chapter 6 - Queensland Planning System: Environmental Performance Evaluation, Part II

6.1 Introduction

This chapter concludes the second part of the thesis by examining the effectiveness of the Queensland Planning System in achieving biodiversity protection outcomes. The previous chapter provided an analysis of the overarching institutional setting which determines the extent and nature of the regulatory responses to the biodiversity protection problem in Queensland. It revealed that the Sustainable Planning Act 2009 (Qld) (SPA) forms only part of a complex institutional framework, which creates both opportunities and constraints to the protection and sustained use of ecological systems and biodiversity.

This chapter analyses the SPA as a separate sub-system of rules regulating a wide range of development activities in Queensland, with particular focus on vegetation clearing. The chapter consists of four parts. It starts (section 6.2) with a description of the methodological approach. The second part (section 6.3) examines the historical context of the SPA. The third (sections 6.4 and 6.5) describes the main elements of the system, their content and regulatory role. In particular, it examines the environmental regulations of the SPA, which incorporate provisions of the Vegetation Management Act 1999 (Qld) (VMA) (the SPA-VMA framework). The final part (section 6.6) discusses the strengths and weaknesses of the environmental performance of the SPA-VMA framework and its future trajectory.

6.2 Methodological approach

The overall methodological approach taken to this study and its limitations has been described in the previous chapter (see section 5.2). This section describes in more detail the approach related to the analysis of the SPA-VMA framework. It describes the structure, scope and timelines of the study. This section also contains a brief discussion of an alternative approach initially selected for the study and its outcomes.

6.2.1 Structure of the study

The SPA is a complex, hierarchical and ‘nested’ system. The overall framework comprises various sub-sets of rules such as the Sustainable Planning Regulation 2009 (Qld) (SPR), state and local planning instruments, and development assessment codes and policies. The SPA also provides for the integrated development assessment system (IDAS), which links several development assessment frameworks of other statutes, including the VMA. The structure of the regulatory framework and its elements is outlined in Figure 18.
To indicate the level of complexity of the system, it is worth noting that the SPA is 732 pages long, supported by 247 pages of regulations. Furthermore, there are 11 statutory regional plans, and development in each local government area is regulated under a separate local planning scheme. Currently there are 11733 local government planning schemes in Queensland (DSDIP 2014a), which differ considerably in the scope and detail of regulation. For example, the Redlands Planning Scheme is a 1910 page long document. In contrast, the planning scheme of Wujal Wujal Aboriginal Shire comprises 149 pages. This system is complemented by development assessment codes designed to achieve the purposes of other statutes integrated through the IDAS.

The SPA structures interactions among diverse institutional actors ranging from individual developers to the state governmental agencies responsible for the development and implementation of rules and monitoring of compliance. The regulatory system incorporates numerous decision-making ‘arenas’ which produce outcomes and outputs having relevance for ‘on ground’ biodiversity protection outcomes. The complexity of the system exceeds the capacity of a single study to address all or even the majority of rules in any detail. Therefore, to scope the rules for the analysis, this study follows two major steps. First, it selects the focal decision-making situation (i.e. ‘arena’). Second, it selects several ‘critical’ clusters of rules affecting the situation and its outcomes.

The application of the IAD framework, and consequently the diagnostic framework of this study, requires selection of the focal action arena(s). The IAD framework does not prescribe their number, and the analysis can be conducted based on either a single or multiple decision-

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33 Note: the number of schemes exceeds the number of councils (77) due to the amalgamation reform in 2009 (see chapter 5)
making situations (Ostrom 2005). To select the focal situation this study applies the so called ‘critical case’ approach (Patton 1990, Flyvbjerg 2001). In the methodology, a critical case is defined ‘as having strategic importance in relation to the general problem’ (Flyvbjerg 2001:79). Thus, the decision-making situation selected for analysis should validate claims regarding the environmental performance of the whole system, as per established boundaries.

Drawing on the ‘critical case’ approach, the assessment was based on the rules regulating development assessment decisions (see Figure 18). This focal situation can be described as a critical determinant of the environmental performance of the SPA, because this is where decisions, such as development approval or conditional approval, authorise the actions which, if undertaken, will affect the state of the biophysical environment. Therefore, it indicates the extent to which the system can potentially prohibit or limit adverse development effects on the natural environment and biodiversity. Furthermore, it indicates the ‘cumulative’ effect of different interacting layers of rules operating within the system. The choice of this arena, however, does not imply a single set of actors, as development assessment in Queensland can be conducted by both state and local governments.

The second step involved selection of the main groups of rules which affect the decision-making situation. Similarly to the analysis of the broader regulatory framework conducted in chapter 5, the scoping of rules was guided by the set of questions provided in Table 13, which are linked to different categories of rules of the diagnostic framework.

**Table 13 Guiding questions for data scoping, Part II**

<table>
<thead>
<tr>
<th>Number</th>
<th>Question</th>
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<tbody>
<tr>
<td>1</td>
<td>What is the overarching purpose of the regulatory framework? What are the functional and geographical scales? What environmental outcomes the system undertakes to achieve? How environmental outcomes are positioned (balanced) against conflicting outcomes?</td>
</tr>
<tr>
<td>2</td>
<td>Which institutional actors are responsible for the regulatory decisions determining environmental outcomes within the system?</td>
</tr>
<tr>
<td>3</td>
<td>What regulatory instruments have been established? How do they contribute to the achievement of environmental outcomes? Which instruments are the crucial determinants of biodiversity protection outcomes?</td>
</tr>
<tr>
<td>4</td>
<td>What measures have been adopted to coordinate different regulatory authorities and resolve conflicting matters? Which regulatory authority has control over the final decision?</td>
</tr>
<tr>
<td>5</td>
<td>What information systems have been established to gain understanding of the state of biodiversity and development impacts? How the state is monitored and reported?</td>
</tr>
<tr>
<td>6</td>
<td>What incentives/disincentives established system provides for the regulators to pursue biodiversity protection outcomes?</td>
</tr>
</tbody>
</table>

Source: author

Data analysis was based on a qualitative approach and followed the structure of the diagnostic framework developed in chapter 4 (Table 5). Diagnostic questions guiding the analysis are incorporated in the discussion section of this chapter (section 6.6). It should be noted that the scope of the analysis covers five performance dimensions: scope, choice, position-boundary, information and payoff rules. Aggregation rules have not been covered separately, as the object of the analysis is an integrated system, where the final decision-making power is concentrated in
a single government agency. Therefore the analysis of position-boundary rules covers aspects of the ‘aggregation’ dimension.

6.2.2 Scope of the study

The SPA provides for the regulation of a wide range of development activities which can be causally linked to the changes in the biophysical environment. As a result, biodiversity protection outcomes can be linked to diverse regulatory measures, including those limiting the rate of environmental degradation such as restricting the release of contaminants or the exposure of acid sulphate soils. Such a broad formulation of the problem, however, can produce confusing results, as the reduction in the rate of adverse environmental effects does not necessarily result in improvement in biodiversity.

To examine regulatory responses to the loss of terrestrial biodiversity, this chapter considers a relatively narrow scope of regulations embedded in the SPA framework. It covers only the rules regulating development activities that result in vegetation clearing. Vegetation clearing is most directly linked to habitat loss and fragmentation and, consequently, the loss of terrestrial biodiversity. Furthermore, clearing in riparian and coastal zones also contributes to bank erosion and nutrient runoff which affect aquatic systems (see section 5.2.3).

Within the SPA framework there are two major sub-sets of instruments that regulate vegetation clearing and guide the two regulatory authorities involved in development assessment. The first is a system consisting of land use zone and overlay regulatory maps, and associated assessment codes, which are incorporated in local government planning schemes. It regulates development assessment in a local government area and applies to local government as assessment manager. The second system consists of regulatory vegetation maps and associated assessment requirements developed under the VMA and integrated in the SPA through the IDAS (see Figure 18). It has a state-wide application and guides decisions of the responsible agency of the state government. Local governments are required to reflect these provisions in their planning schemes.

This chapter focuses only on the SPA-VMA framework. Local government planning schemes have been excluded from the scope of analysis for several reasons. First, as established in chapter 5, vegetation management in Queensland is a ‘state interest’ regulated by the state government. The Queensland government prepares regulatory maps and associated development assessment instruments, which are applicable across the state. Second, despite authorisation, local governments encounter significant disincentives to undertake environmental regulation (see section 5.7.6 for a discussion). In practice, they are constrained in their abilities to add another vegetation protection ‘layer’ to the system, apart from that authorised by the state government. Therefore, while locally important, the additional provisions included in some local government planning schemes would not significantly alter biodiversity protection
outcomes at the state scale. Furthermore, aggregating data from 117 planning schemes, most of which have been adopted under the previous Integrated Planning Act 1997 (Qld) (IPA), would be very complicated and time consuming. Finally, most of the local planning schemes that regulate urban pressures in the coastal zone, and would be of interest for some additional analysis, are currently under the review. According to the Department of State Development and Infrastructure Planning, the new planning schemes will reach the adoption stage only in the period from mid-2014 to mid-2015 (DSDIP 2014c).

6.2.3 Assessment timelines

This part of the analysis takes a different approach to the selection of temporal points of reference for the study. Chapter 5 identified significant problems with the ‘regulatory turn’ taken by the LNP Queensland government. This chapter therefore concentrates on the recent 2013 regulatory reforms and examines to what extent they have affected the potential of the SPA-VMA framework to contribute to the protection of biodiversity. Based on the analysis of documentary sources, it discusses the future trajectory of the biodiversity protection in the state under these regulatory reforms. All regulatory provisions examined in this chapter are current as at May 2014, if not indicated otherwise.

6.2.4 Evaluation protocols and quantitative analysis: lessons learned

As indicated previously, many parts of the research design were changed during the course of the study. This section concludes with a description of the analytical approach initially intended for the analysis of the SPA. It is included here to outline the methodological challenges encountered in institutional analysis. The aim is not to criticise a particular approach, but to provide information on the factors that researchers need to be aware of in the analysis of selected sub-systems of rules.

The initial approach taken to data gathering and analysis was inspired by the studies of Brody (2003), Brody and Highfield (2005) and the work of other scholars (e.g. Berke et al. 1996) which examined the quality of land use plans. Brody (2003) studied the ability of local comprehensive plans in Florida, United States, to incorporate ecosystem management principles. Based on theoretical conceptions of plan quality he evaluated local plans against a conceptual model of a high quality ecosystem plan. Brody (2003) conceptualised and measured five plan components: factual basis, goals and objectives, inter-organisational coordination and capabilities, tools and strategies and implementation. A set of indicators was provided to disaggregate each component. The analysis was supported by a ‘plan coding protocol’.

This study attempted to follow the above research process. As the planning systems in the U.S. studied by Brody (2003) were outcome-based in their approach, adapting the same protocol was

34 Note: this approach differs from what was originally intended to make the study more relevant to the current situation in Queensland and more useful as a basis for further research.
considered beneficial for potential comparative purposes. The protocol was adjusted to suit the regulatory scope and structure of the local planning instruments developed under the IPA/SPA, and three elements of the instruments were selected: the hierarchy of goals and objectives (i.e. desired environmental outcomes and outcomes for ‘environmental’ zones and overlays), regulatory maps (zones and overlays), and planning policies and schedules. Each element was examined to determine whether it addressed ‘environmental’ matters and how well the matters were specified. The collected data was expected to determine the ‘environmental performance’ of this instrument and, consequently, that of the Queensland Planning System.

The research process started with the selection of a sample of instruments - initially all coastal local governments based on the level of development pressures. It then narrowed down the group of instruments using eight criteria: population size, density, growth and growth projection, annual residential and industrial building value, annual operating income and full time staff employed. A scoring approach allowed grouping them into five categories to account for potential differences in analytical outcomes. Four categories were selected for the analysis which included 15 local governments operating 25 pre-amalgamation coastal planning schemes.

Difficulties emerged with the interpretation of data. All protocols clearly indicated that, compared to the standards set out for the plan quality, Queensland local government planning schemes are very ‘basic’ in addressing such complex problem as biodiversity loss. The importance of biodiversity protection was acknowledged at the strategic level; however, regulatory maps contained little detailed information. Similarly, with a few exceptions for particular values such as koala habitats, ‘acceptable solutions’ incorporated in assessment codes lacked specificity. The study, thus, led to the academically ‘correct answer’ that the plans were ineffective. However it also raised several questions: Can local government planning schemes serve as the basis for measuring effectiveness of the Queensland Planning System? Can instruments be ‘taken out of the system’ and analysed separately? Who establishes ‘quality’ criteria? And finally: Can such a study offer a platform for policy recommendations or reforms?

Detailed examination of these questions led to two outcomes: first, a more critical approach to the ‘generic’ criteria and propositions made in the academic literature; and second, a re-examination of the scope and structure of the study. The variety of academic opinions, and weaknesses of the ‘generic’ principles and standards for institutional design, were examined in chapter 3 and some reflections are included in chapter 7. They will not be repeated here. However, these necessitated a re-examination of the scope of the study, which led to two outcomes. First, the scoring approach and detailed examination of a single instrument was abandoned in favour of a more comprehensive, but less detailed, diagnostic approach. Second,
the analysis was redirected to search for ‘critical’ interacting clusters of rules within the whole system.

These changes in the analytical approach also affected the outcomes of the study. For example, examination of the whole regulatory framework allowed identifying that biodiversity protection in Australia is a matter of national and state interest. In this governance system, ‘environmental’ planning (e.g. remnant vegetation mapping, design of protected area networks) is a separate process carried out at a different level. Moreover, allocating greater regulatory responsibility to local governments would not necessarily be an improvement. Queensland local governments have resource capacity problems and generally are not motivated by the system to address biodiversity protection matters.

In the context of institutional analysis, three lessons were derived from the experience with fixed assessment designs. First, each governance system is different. Therefore, the analysis of particular sub-system of rules requires detailed understanding of its role in the overall regulatory system. The ‘weakness’ of a particular regulatory instrument may indicate three things: ineffectiveness of the regulatory response to the problem, absence of the problem (i.e. no need for regulation), or the presence of other instruments addressing the problem. In other words, analysing selected instruments as ‘stand-alone’ systems runs the risk of misinterpretation. Second, a clear understanding is required of the power relation structure, and the capacities and interests of involved governance actors. General assumptions that ‘local governments’, ‘communities’ need to be ‘environmental’ regulators or planners can lead to misleading conclusions. Finally, the use of preselected sets of indicators and criteria should be approached with caution. Rules are developed in response to a particular problem and fitted into a particular social and institutional context. What makes an ‘effective’ or ‘ineffective’ response depends on many factors which can be uncovered only through a qualitative analysis of the system.

To conclude, this discussion does not dismiss the applicability of pre-established quantitative evaluation design to institutional analysis. If correctly designed, it can provide a good insight into diverse aspects of the problem and operate as another ‘diagnostic’ tool along with qualitative evaluation. Limitations of a single approach, and the need to combine both qualitative and quantitative methods, are also acknowledged by Brody and Highfield (2005). However, whether the results obtained through quantitative analysis are worth the time and effort at such complex governance levels is another question.

6.3 Integrated planning in Queensland: historical background

In Queensland, institutionalisation of environmental considerations in land use planning and development assessment has an overall history of less than three decades. To better understand the structure and operation of the current system, this section briefly outlines the history of these
regulations. In particular, it focuses on the major reform in the planning system introduced under the *Integrated Planning Act 1997 (Qld)* and encountered challenges.

**Integrated Planning Act**

The first statute providing for land use planning in Queensland, the *Local Government Act 1936 (Qld)*, covered regulation of development activities (Leong 2002/2003). Rules incorporating environmental matters appeared only in 1990 with the *Local Government (Planning and Environment) Act 1990 (Qld)* (PEA). One of the objectives of the PEA was ‘to provide a code by which a local government or the Minister may undertake the planning of an area to facilitate orderly development and the protection of the environment’ (PEA, s.1.3 (a)). The PEA defined the concept of ‘environment’ and introduced the environmental impact statement as a tool to manage development effects (Leong 2002/2003). However, as England (1999:125) argues, in the PEA ‘environmental protection was an adjunct to the core objective of orderly town planning’. Both statutes were based on prescriptive zone planning, where zoning determined the permitted type of development (Hopewell 2002).

The *Integrated Planning Act 1997 (IPA)* was a comprehensive reform of the Queensland planning and development assessment system. Its major purpose was to improve economic performance and competitiveness of Queensland due to ‘globalisation pressure’ (Yearbury 1998). The IPA followed the major public sector reforms in Australia, which led to improvements in economic efficiency of the public sector, re-distribution of government services between public and private sector, as well as adoption of strategic management techniques (England 2004). As Yearbury (1998:200), then Director General of the Queensland Department of Local Government and Planning, stated the IPA ‘is designed to deliver substantial benefits in terms of creating a better business climate for investing in Queensland while at the same time requiring environmental conduct commensurate with community expectations.’ The new framework was expected to implement indicators of a high performance planning and development assessment system: simplicity, clarity, transparency, responsiveness, accountability and customisation (Yearbury 1998).

Along with other reforms, the IPA brought significant changes in environmental regulation. It introduced the concept of ‘ecological sustainability’ as an overarching purpose. The concept was defined as a balance that integrates environmental, economic and social dimensions (IPA, s1.3.3). The environmental dimension was defined as ‘protection of ecological processes and natural systems at local, regional, state and wider levels’ (IPA, s1.3.3 (a)), which occurs if ‘the life supporting capacities of air, ecosystems, soil and water are conserved, enhanced or restored for present and future generations and biological diversity is protected’ (IPA, s1.3.6 (a)). The IPA also required the application of a precautionary principle, as well as accounting for short and long-term environmental effects of development at local, regional, state and wider levels (IPA, s1.2.3. (1) (a) (ii) and (iii)).
Influenced by New Zealand’s Resource Management Act 1991 (Spiller 2003), the most important reform of the IPA was a shift from prescriptive zone planning to strategic or performance based planning. Instead of regulatory zoning, planning schemes were premised on the achievement of strategic outcomes (England 2004). The IPA defined only ‘key elements’ which must be addressed by local planning schemes, among them: any state and regional dimension, including ‘resources or areas of ecological significance’ (IPA s2.1.3A (1) (c) and (4)), identified desired environmental outcomes, measures that facilitate their achievement, and a priority infrastructure plan36 (IPA s2.1.3 (1)). This approach allowed for greater discretion in determining land use while assessing impacts of proposed development through performance criteria (Baker et al. 2006). Consequently, local governments had a high discretion in determining the content of the planning schemes (England 1999).

To streamline and simplify the development assessment process, the IPA introduced the Integrated Development Assessment System (IDAS) for ‘integrating state and local government assessment and approval processes for development’ (IPA, s3.1.1). A broad definition of ‘development’ allowed creating a decision-making system that integrates assessment of a large variety of development activities regulated under other statutes. The vertical integration was pursued by including regional and state level planning matters into local government planning schemes ‘so that development opportunities and standards are more readily identifiable’ (Explanatory Notes, IPB 1997:3).

Despite promised improvements, the implementation of this planning reform encountered several challenges. Problems emerged with the application of the performance-based approach. Local governments experienced difficulties with drafting new performance-based planning schemes. As a result, in 1999 the Queensland government issued a range of guidelines and notes on the design of the schemes. However, insufficient guidance on the structure and terminology led to a variety of designs which were criticised as adding to the overall complexity of the system (O’Hart 2007/2008). To simplify the schemes, several changes have been made in the IPA. For example, initially, among its ‘key elements’ the IPA also included performance indicators to assess the achievement of the overall outcomes set out for the planning scheme. In 2003, however, this performance evaluation element was abandoned.

Other elements of the IPA design were also subject to criticism. The overall framework was considered as being too complex and suffering from ‘legislative obesity’ (Fogg 2004/2005), as well as failing to effectively incorporate satellite legislation (Welford 2003/2004, Walton 2006/2007). It was also argued that the IPA’s focus on the process was negatively affecting the quality of planning outcomes, as planners were overwhelmed with process management and distracted from their core task (Cumming 2001, Leong & Smith 2003/2004, Reid & Fjeldsoe 2007). The concept of ‘ecological sustainability’, as an overall purpose of the IPA, was also

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36 Note: in 2003 priority infrastructure plan replaced initial requirement to include benchmark development sequence.
criticised as being too complex and indeterminate (England 1999). Both decision-makers and courts were facing problems in interpreting and applying this standard in practice (Walker 2002/2003). The implementation was also hindered by capacity problems, as local governments experienced shortages in both financial resources and qualified staff (Shaw 2002, Leong & Smith 2003/2004, Wypych et al. 2005). Preparation of a planning scheme for a medium sized council costed around $1.5 million (DLGPSR 2007).

Several problems were identified in the design of local planning instruments. They became more complex and suffered from internal inconsistencies (Wypych et al. 2005, England 2006). The need to integrate state and regional dimensions also contributed to the overall complexity and uncertainty (Fogg 2004/2005, Walton 2006/2007). Performance based codes, which were crucial for assessing proposed development, were criticised for weak consistency between different outcomes and criteria, as well as for poorly defined performance standards (Wypych et al. 2005, Roughan 2007). Councils were also failing to determine specific and consistent environmental outcomes (Reid & Fjeldsoe 2007). Particularly significant concerns were expressed with regard to the regulation of environmental issues. England (1999:139), examining provisions of the IPA, concluded that ‘there is little in the Act that actually makes planning law in Queensland any greener than before.’ Similarly, the Attorney-General Rod Welford (2003/2004:60) concluded that despite greater environmental awareness ‘evidence that the environmental component of planning, as distinct from more traditional planning principles, is receiving significantly higher priority or appreciation in planning decisions than previously is mixed’. Furthermore, Summers (2007) argued that, despite improvements in information, strategic biodiversity outcomes were not realised due to conflicting outcomes in planning schemes.

**New Reform: Sustainable Planning Act 2009**

In 2006, the Queensland government initiated an extensive review of the IPA performance. On 23 August 2007 the Government released the report *Planning for a Prosperous Queensland, A reform agenda for planning and development in the Smart State* (Reform Agenda) (DLGPSR 2007). The Reform Agenda aimed ‘to move the focus from the planning process to the delivery of sustainable outcomes’ (DLGPSR 2007:1). It consisted of 80 key actions covering different management areas of the IPA and IDAS. The reform focused on improvement in tools, proactive management of development, reduced complexity through standardisation, adoption of a risk-based approach to development assessment, streamlined dispute resolution processes, and improved community participation. However, the evaluation of the achievement of ‘ecological sustainability’ did not play a significant role in the review (O’Hart 2007/2008).

The Reform Agenda explicitly acknowledged challenges with preparation of local planning schemes and related delays. It concluded (DLGPSR 2007:8):
Many planning schemes prepared under the IPA are lengthy and complex, and demonstrate scope for far greater standardisation of key elements (e.g. definitions). The lack of prescription under the IPA complicates the plan-making process with substantial resources expended on matters such as format, structure and definitions. Such work distracts from strategic analysis and drafting outcomes which respond to local issues. Detailed knowledge of the framework of individual schemes is required to understand their provisions. This complicates the interpretation of planning schemes, creates uncertainty for the end user, and limits the ready transfer of knowledge across local government boundaries.

The reform resulted in the new Sustainable Planning Act 2009 (SPA) which came into effect on 18 December 2009. The change of the title of the Act was made ‘to reflect a stronger focus on achieving ecological sustainability’ (Explanatory Notes, SPB 2009). To improve vertical and horizontal integration and consistency, the SPA clarified the hierarchy between the state planning instruments and introduced standard planning schemes (Explanatory Notes, SPB 2009). The SPA also increased the role of the state government by expanding the Minister’s powers to direct local governments to make or amend a local planning instrument. The major amendments concerned the IDAS and were focused on ‘fast tracking’ development assessment.

Despite the reform, previously adopted local government schemes remain in force. The SPA explicitly provided that a local government’s planning scheme ‘made under repealed IPA that is in force immediately before the commencement continues to have effect and is taken to be the planning scheme for the local government’s planning scheme area made under this Act’ (s778(1)).

### 6.4 Sustainable Planning Act: regulatory structure

This section examines the design of the SPA. It describes the major instruments of the system, their hierarchy and the IDAS provisions (for a detailed analysis of the SPA see England 2011). This description is provided to assist with understanding of the instruments accounting for ‘environmental performance’ of the SPA-VMA framework examined further in this study.

#### 6.4.1 Regulatory scope

As in the IPA, the overarching purpose of the SPA is ‘ecological sustainability’. It is defined as ‘a balance that integrates the protection of ecological processes and natural systems at the local, regional, state and wider levels, economic development and the maintenance of the cultural, economic, physical and social wellbeing of people and communities’ (s8). The SPA seeks to achieve its purpose by (s2):

(a) managing the process by which development takes place, including ensuring the process is accountable, effective and efficient and delivers sustainable outcomes; and
(b) managing the effects of development on the environment, including managing the use of premises; and
(c) continuing the coordination and integration of planning at the local, regional and state levels.

The SPA requires any entity conferred with a function or power under the Act to ‘perform the function or exercise the power in a way that advances this Act’s purpose’ (s4(1)(a)). Exception
is made only for assessment managers and concurrence agencies ‘other than a local government’ which ‘must have regard’ to the Act’s purpose (s4(1)(b)(c)). This requirement also does not apply to code or compliance assessment decisions made under the Act (s4(2)).

The term ‘environment’ is defined in the SPA very broadly and includes (schedule 3):

(a) ecosystems and their constituent parts including people and communities;
(b) all natural and physical resources; and
(c) the qualities and characteristics of locations, places and areas, however large or small, that contribute to their biological diversity and integrity, intrinsic or attributed scientific value or interest, amenity, harmony, and sense of community; and
(d) the social, economic, aesthetic and cultural conditions affecting the matters in paragraphs (a), (b) and (c) or affected by the matters.

The regulatory scope is very broad and is defined by the term ‘development’, which specifies the type of activities regulated under the Act. Many different types of development regulated under other statutes are incorporated through the IDAS. Section 7 of the SPA distinguishes five broad groups of regulated development activities:

(a) carrying out building work;
(b) carrying out plumbing or drainage work;
(c) carrying out operational work;
(d) reconfiguring a lot;
(e) making a material change of use of premises.

The scope is more precisely defined in section 10. For example, ‘carrying out operational work’ includes (emphasis added):

- extracting gravel, rock, sand or soil, conducting a forest practice, excavating or filling, placing an advertising device, undertaking work that materially affects premises or their use, clearing vegetation, undertaking operations that allow taking or interfering with water, tidal works, works in a coastal management district, waterway barrier works, work in a declared fish habitat area, removing, destroying or damaging a marine plant, roadworks on a local government road.

Listing of activities as such, however, is not the trigger for development assessment under the SPA. Only those developments prescribed as ‘assessable development’ require approval. Assessable development is further specified in the Sustainable Planning Regulation 2009 (Qld) (SPR) and local government planning schemes.

### 6.4.2 Planning instruments

The SPA establishes the hierarchy of state and local planning instruments which together form a toolset for advancing the Act’s purpose. This section briefly addresses each group of planning instruments, their role, content, legal status, and preparation requirements.

**State planning instruments**

A state planning instrument is prepared to articulate the position of the Queensland government on planning- and development-related matters of state interest. Under the SPA (Schedule 3) ‘state interest’ is defined as (emphasis added):
There are four types of state planning instruments that can be used, either individually or in combination, to address relevant matter. They can be developed to regulate the matter in the state or its part (SPA s15):

- a State planning regulatory provision;
- a State planning policy;
- a regional plan;
- the standard planning scheme provisions.

A State planning regulatory provision (SPRP) is an instrument which can be prepared to regulate development and to set out mandatory requirements, including development restrictions. A SPRP can be made for restricted purposes (s16):

- to provide regulatory support for regional planning,
- to provide for a charge for the supply of infrastructure, and
- to protect planning scheme areas from adverse impacts.

A SPRP is a statutory instrument and has the force of law (s17). The power to make a SPRP is allocated to the Minister responsible for administering the SPA (the Planning Minister). The SPRP can also be made jointly with another eligible Minister. In case of inconsistency a SPRP prevails over any other planning instrument (s19).

Currently, there are seven SPRPs issued to regulate different development activities in particular localities. Of particular importance are: the *South East Queensland Regional Plan 2009-2031 State Planning Regulatory Provisions* (SEQ Regional plan SPRPs) (DSDIP 2014d), and the *South East Queensland Koala Conservation State Planning Regulatory Provisions* (SEQ Koala Conservation SPRPs) (DERM 2010). The SEQ Regional plan SPRP supports the implementation of the SEQ Regional plan and sets the boundaries for urban footprints in the SEQ Region. The SEQ Koala Conservation SPRP aims to protect koala population in identified habitat areas. They specify development that is prohibited and provide for an additional set of assessment criteria for code- and impact-assessable development.

A State planning policy (SPP) aims to advance the purpose of the SPA ‘by stating the State’s policy about a matter’. It allows setting out policy direction and expressing desired planning and development outcomes of state interest for a single or several matters. It may also include detailed codes for development assessment. A SPP is a statutory instrument (s23) and in the hierarchy prevails over a regional plan and local planning instruments (s25). A SPP can be made by the Planning Minister or jointly with the eligible Minister (s26). It can be in effect for 10 years unless repealed earlier (s27(1)).

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37 Note: the Cabinet structure and portfolio distribution is described in chapter 5.
SPPs must be addressed through local government planning schemes, regional plans, and when making decisions about designating land for infrastructure (DSDIP 2013c). If there is a discrepancy between a local government planning scheme and the SPP, the SPP overrides the planning scheme. Historically, there has been a practice to issue a SPP for a particular scope of regulatory matters such as coastal development, air, noise pollution and similar. However, since December 2013 all policy matters have been integrated in a single State Planning Policy which is a central instrument incorporating state interests in land use planning (DSDIP 2013c). The State Planning Policy has effect throughout the entire state of Queensland.

The matters of the State Planning Policy include 16 state interests, which are presented under five broad themes: liveable communities and housing, economic growth, environment and heritage, and hazards and safety (DSDIP 2013c)38. The Policy is also supported by an interactive mapping system published and updated by the DSDIP (www.dsdip.qld.gov.au/spp-mapping). Priorities of interests, however, are not defined. As stated in the document (DSDIP 2013c:10):

The SPP does not prioritise one state interest over another at a statewide level. It acknowledges that the manner in which state interests need to be applied will vary between and within regions and local government areas, depending on many different environmental, economic, cultural and social factors.

Regional plans integrate state interests for a geographic area. They advance the purpose of the SPA ‘by providing an integrated planning policy for the region’ (s33). The key elements of regional plans are the desired regional outcomes, policies and actions for achieving those outcomes, and the desired future spatial structure for the region (s38). Regions are established under a regulation that prescribes the local government areas or their parts as a designated region (s32). The plan is a statutory instrument (s34) and, in case of inconsistency, prevails over a local planning instrument (s36). As noted above, regional plans can be supported by SPRPs.

The responsibility for making a regional plan is allocated to ‘the regional planning Minister’ (s37). The Minister is required to establish a planning committee for the region, which must include mayors or councillors of the local governments of the region, and persons with appropriate qualifications, experience or standing (s43). The committee has only an advisory role.

Regional land use plans, as statutory instruments, first emerged in 2005 with the adoption of the South East Queensland Regional Plan 2005-2026. Currently, there are 11 statutory regional plans in Queensland. They differ in their scope and matters covered. In this context a distinction should be made between the ‘old’ (pre 2012) and ‘new’ generation plans. The ‘old’ plans cover a broad scope of policy matters (e.g. Far North Queensland Regional Plan), as historically this instrument was situated ‘above’ SPPs. The ‘new’ plans (e.g. Darling Downs Regional Plan and Central Queensland Regional Plan) focus on a limited scope of matters. As stated in the State Planning Policy (DSDIP 2013c:8):

38 Note: ‘Biodiversity interests’ will be examined in more detail in section 6.5.
It is not necessary for a regional plan to provide regional policy about all state interests. The focus will typically be on those matters requiring specific regional guidance as determined to be relevant or necessary by the planning Minister.

Standard planning scheme provisions (SPSPs) provide the means to standardise various aspects of future planning schemes. They advance the purpose of the SPA by providing for a ‘consistent structure for planning schemes and standard provisions for implementing integrated planning at the local level’ (s50(b)). The SPA authorises the Planning Minister to make standard planning scheme provisions ‘for the whole of State’ (s54). SPSPs are a statutory instrument and have the force of law (s51), but they do not regulate or affect development, but prevail to the extent of inconsistency (ss52, 53).

The *Queensland Planning Provisions version 3.0* (QPP) (DSDIP 2013d) are the standard planning scheme provisions made under the SPA. The QPP comprises two modules that contain mandatory and optional components:

- Module A incorporates the template that each local government ‘must’ adopt for its planning scheme which includes such elements as overall structure, list of planning scheme components, mapping standards, assessment code templates, definitions;
- Module B provides guidance about the drafting of all elements and identifies the scope of optional information that may be included by a local government.

Currently, the instrument affects only a limited number of local government planning schemes, as it does not apply to the old IPA schemes (s55A). The bulk of new planning schemes will be adopted in 2015.

The SPA establishes a single legislative process for the making, amending, and repealing of all state planning instruments. Public notification and consultation is a statutory requirement. The consultation periods depend on the instrument and range from 30 to 60 business days (see s60). The preparation process of state planning instruments is also regulated by the guidelines of the Cabinet (DPC 2013).

**Local planning instruments**

The SPA (s77) provides for three types of local planning instruments: a planning scheme, a temporary local planning instrument, and a planning scheme policy. A temporary local planning instrument and a planning scheme policy play a minor role in regulation and therefore will not be examined in detail here.

The central instrument of the SPA is the local government planning scheme. It advances the purpose of the Act ‘by providing an integrated planning policy for the local government’s planning scheme area’ (s79(b)). The scheme provides for detailed land-use and development regulation at the property level. It is a statutory instrument and has a force of law (s80). The planning scheme applies to all of the local government’s area (s82(1)).
The SPA prescribes key elements for the planning schemes. It requires that a local government and the Minister ‘must’ be satisfied that the local government’s planning scheme (s88(1)(a)) (emphasis added):

(a) appropriately reflects the standard planning scheme provisions;
(b) identifies the strategic outcomes for the planning scheme area;
(c) includes measures that facilitate achieving the strategic outcomes;
(d) coordinates and integrates the matters, including the core matters, dealt with by the planning scheme, including any State and regional dimensions of the matters;
(e) includes a priority infrastructure plan.

The content of the planning schemes is specified under the ‘core matters’. The SPA distinguishes three groups of core matters for the planning scheme: land use and development; infrastructure, and valuable features (s89(1)). In environmental matters, ‘valuable features’ include ‘each of the following’ (s89(2)) (emphasis added):

(a) resources or areas that are of ecological significance (e.g., habitats, wildlife corridors, buffer zones);
(b) areas contributing significantly to amenity (e.g., areas of high scenic value);
(c) areas or places of cultural heritage significance (e.g., aesthetic, architectural, indigenous cultural significance);
(d) resources or areas of economic value (e.g., extractive deposits, fishery resources, forestry resources, water resources, agricultural land).

The power to make a planning scheme is allocated to a local government (s84). As these instruments integrate both state and local matters of interest, their adoption is time- and resource-consuming. As the DSDIP (2014c) schedule suggests, it may take 2-3 years for a draft to be finalised. As a result, this instrument is the most ‘stable’ element in the whole system which even survives the changes of the overarching statutory framework such as the IPA.

6.4.3 Development assessment system

Categories of development

The second major element of the SPA is the Integrated Development Assessment System (IDAS) which integrates the state and local government assessment and approval processes for development (SPA s230). The SPA establishes five categories of development (s231) (emphasis added):

- ‘exempt’ development means that the development needs not to comply with planning instruments, other than a State planning regulatory provision (s235(2)). Under the SPA all development is exempt unless it is not assigned to other four categories (s231(2));
- ‘self-assessable’ development means that development permit is not required. However, the development must comply with applicable codes. Development in contravention of applicable codes is an offence (s236);

Note: as this chapter does not focus on local government planning schemes, it will not expand on design details here (see QPP for a structure).
development requiring ‘compliance assessment’ means that development, documents or works are assessed for compliance with a matter prescribed under a regulation, a planning instrument or a condition of a development approval (s393). A compliance permit is necessary (s237(2));

‘assessable’ development means that assessment is required under the IDAS by authorised government entity. There are two types of ‘assessable’ development:

- ‘impact assessable’ means that the application is assessed against the whole of the planning scheme (if assessed by a local government) or against relevant laws and policies (if assessed by the State government) and must be publicly notified. The public has a right to make submissions and appeal a decision.
- ‘code assessable’ means that the application is assessed only against any relevant code (e.g. vegetation clearing code). It is decided by the assessment manager and no public submission or appeal rights exist.

‘prohibited’ development means that a development application cannot be assessed. Local governments are not authorised to prohibit development. Development can be taken to be ‘prohibited’ if it is identified in Schedule 1 of the SPA, a SPRP or SPSP.

The IDAS applies to ‘assessable’ development, which defines the scope and boundaries of the state and local government regulators in relation to development assessment. Characterisation as either ‘impact assessable’ or ‘code assessable’ depends on provisions of the relevant planning scheme and schedules incorporated in the Sustainable Planning Regulation 2009 (Qld) (SPR). The SPR also incorporates a list (Schedules 6 and 7) identifying which agency is responsible for development assessment, and what decision-making sources (codes, laws, policies, prescribed matters) are applicable.

The government entities involved in the IDAS process are referred to as the ‘assessment manager’ and ‘referral agency’. The assessment manager for an application administers and decides the application, but may not always assess all aspects of development (s247). In deciding the application the assessment manager must either (s324):

(a) approve all or part of the application; or
(b) approve all or part of the application subject to conditions decided by the assessment manager; or
(c) refuse the application.

There are two types of ‘referral agencies’ which imply different levels of influence on the decision-making process: ‘concurrence agency’ and ‘advice agency’ (s252). The status of ‘concurrence agency’ means that the entity has the power to refuse the application, or to impose mandatory conditions on a development permit. The status of ‘advice agency’ means that the entity may offer advice to the assessment manager, but it cannot impose mandatory conditions or refuse an application.
The assessment manager can be a local government or a state government agency, if assessment involves state interest. Since 1 July 2013, the Department of State Development, Infrastructure and Planning (DSDIP) is the single lodgement and assessment point for all development applications involving matters of ‘state interest’. All development assessment requirements are integrated in one document The State Development Assessment Provisions (SDAP) (DSDIP 2014e), which set out the state’s criteria for assessing development applications.

**Decision-making instruments**

Codes are a widely applied instrument under the SPA framework. The design of the codes can take two forms: ‘performance-based’ and ‘prescriptive’. The performance-based codes are the most common form in the SPA, in particular for environmental matters. These codes allow developers to be flexible in choices of solutions. The common elements of performance-based codes are purpose statements, outcomes, and a list of probable solutions describing the ways in which the outcomes could be met. A ‘snapshot’ of Module 8 Native vegetation clearing code (DSDIP 2014e) is provided below as an example (Figure 19).

![Figure 19](image-url)

‘Impact assessment’ under the SPA is a different process to an environmental impact assessment (EIA). In the case of impact assessment, the assessment manager must assess the application against relevant state and local planning instruments (if it is a local government) or against laws and policies that are relevant to the application (if it is a state government agency) (s314). Instead of a formal EIA process the SPA provides for an information request system (s276-281).

If development affects Commonwealth interests, or matters of national environmental significance under the EPBC Act (see section 5.5.1), a separate approval may be required. Chapter 9 of the SPA provides for a formal environmental impact statement process. However, it applies only if the development needs to be assessed under the EPBC Act.
6.5 Queensland Planning System: environmental provisions

While the SPA prescribes assessment categories and processes, the criteria for the assessment of vegetation clearing are incorporated in instruments made under the *Vegetation Management Act 1999* (Qld) (VMA). The VMA is perhaps the most controversial and politically difficult ‘environmental’ Act in Queensland, as it restricts the rights of landholders to use land for agricultural purposes.

Prior to the VMA some control of vegetation clearing was established only on Crown leasehold land under the *Land Act 1994* (Qld). Initially, the VMA filled the gap by establishing vegetation control regimes on freehold tenures. In 2004 the Queensland government introduced major reforms to the regulation by phasing out broadscale land clearing by 31 December 2006. The reform created a common assessment system for vegetation clearing on both freehold and Crown leasehold tenures (McGrath 2011). In 2009, the protection regime was extended to high value remnant regrowth vegetation, which had not been cleared prior 1987. The previous state of environment reports and other sources have linked the SPA-VMA regulatory framework to a significant reduction in vegetation clearing rate in the state (see section 5.7.3 for data).

Since the election of the LNP government, the VMA framework has undergone another set of reforms reversing some of the previously established restrictions, and changing ‘protective scope’ of the SPA-VMA framework. In May 2013, the Queensland Parliament passed the *Vegetation Management Framework Amendment Act 2013* (Qld) (VMFAA) introducing a range of amendments to the vegetation management framework with the objective to (Explanatory notes, VMFAB 2013:1):

- reduce red tape and regulatory burden on landholders, business and government;
- support the four pillar economy – construction, resources, agriculture and tourism;
- maintain protection and management of Queensland’s native vegetation resources.

This section examines the major elements of the SPA-VMA regulatory framework. The first part briefly describes the scope of ‘biodiversity state interest’ as specified in the *State Planning Policy* (DSDIP 2013c). The second part describes the regulatory purpose, scope and underpinning ‘ecological’ framework of the VMA, its instruments, and their role in vegetation management. Particular attention is paid to the VMFAA reform and introduced changes in the regulatory structure.

6.5.1 Matters of state environmental significance

In the SPA framework, the regulatory role of the Queensland government in environmental matters is reflected in the *State Planning Policy* (SPP) (DSDIP 2013c), which is the major instrument for articulating state interests in planning and development assessment. ‘Biodiversity’ is one of the four themes incorporated in the SPP’s module *Environment and Heritage*. The overall state interest in biodiversity matters is described as (DSDIP 2013c:27):
Matters of environmental significance are valued and protected, and the health and resilience of biodiversity is maintained or enhanced to support ecological integrity.

In the introductory part the SPP identifies the uniqueness of Queensland’s biodiversity and places emphasis on a range of services that the environment provides. Apart from conservation, ‘biodiversity interest’ is also linked to such services as ‘food, recreation, materials and energy’. The SPP incorporates informative definition of ‘biological diversity’ as (DSDIP 2013c:26):

the variability among living organisms from all sources (including terrestrial, aquatic, marine and other ecosystems and the ecological complexes of which they are part), at all levels of organisation, including genetic diversity, species diversity and ecosystem diversity.

The SPP requires the planning schemes to appropriately integrate state ‘biodiversity’ interest by (DSDIP 2013c:27):

- considering matters of national environmental significance regulated under the Environment Protection and Biodiversity Conservation Act 1999 (Cth);
- identifying matters of state environmental significance;
- locating development in areas that avoids significant adverse impacts on matters of state environmental significance;
- maintaining or enhancing ecological connectivity;
- facilitating the protection of matters of state environmental significance by requiring development to, in order of priority:
  - avoid significant adverse environmental impacts, and
  - mitigate significant adverse environmental impacts, where these cannot be avoided, and
  - where applicable, offset any residual adverse impacts;
- facilitating a net gain in koala bushland habitat in the SEQ region.

The SPP specifies the list of matters of state environmental significance (MSES), which implicitly delineate state-local jurisdictional boundaries in environmental matters and identify the areas of regulatory interventions. The MSES in Queensland include (DSDIP 2013c, Glossary) (emphasis added):

- protected areas (including all classes of protected area except coordinated conservation areas) under the Nature Conservation Act 1992 (Qld)
- marine parks and land within a ‘marine national park’, ‘conservation park’, ‘scientific research’, ‘preservation’ or ‘buffer’ zone under the Marine Parks Act 2004 (Qld)
- areas within declared fish habitat areas that are management A areas or management B areas under the Fisheries Regulation 2008 (Qld)
- threatened wildlife under the Nature Conservation Act 1992 and special least concern animal under the Nature Conservation (Wildlife) Regulation 2006 (Qld)
- regulated vegetation under the Vegetation Management Act 1999 (Qld) that is:
  - Category B areas on the regulated vegetation management map, that are ‘endangered’ or ‘of concern’ regional ecosystems
  - Category C areas on the regulated vegetation management map that are ‘endangered’ or ‘of concern’ regional ecosystems
  - Category R areas on the regulated vegetation management map
  - areas of essential habitat on the essential habitat map for wildlife prescribed as ‘endangered wildlife’ or ‘vulnerable wildlife’ under the Nature Conservation Act 1992
  - regional ecosystems that intersect with watercourses identified on the vegetation management watercourse map
  - regional ecosystems that intersect with wetlands identified on the vegetation management wetlands map

Note: this definition is included in informative section and it has not the same weight as definitions included in schedules of the statutes. It is derived from Australia’s Biodiversity Conservation Strategy 2010–2030.
regional ecosystems that intersect with wetlands identified on the vegetation management wetlands map
- high preservation areas of wild river areas under the *Wild Rivers Act 2005* (Qld)
- wetlands in a wetland protection area or wetlands of high ecological significance shown on the Map of Referable Wetlands under the *Environmental Protection Regulation 2008* (Qld)
- wetlands and watercourses in high ecological value waters as defined in the *Environmental Protection (Water) Policy 2009* (Qld), schedule 2
- legally secured offset areas.

It should be noted that the new SPP, introduced by the LNP government, has created a regulatory ‘gap’ in protection of threatened wildlife. ‘State interest’ does not include protection of essential habitat for ‘near threatened’ wildlife category protected under the *Nature Conservation (Wildlife) Regulation 2006* (Qld). This category has been also removed from the VMA and its instruments.

### 6.5.2 Integrated vegetation management framework

*Purpose and regulatory scope*

The purpose of the VMA is to regulate the clearing of native vegetation in a way that protects a range of environmental values (s3):

- conserves remnant vegetation and vegetation in declared areas,
- ensures the clearing does not cause land degradation,
- prevents the loss of biodiversity,
- maintains ecological processes,
- manages the environmental effects of the clearing and reduces greenhouse gas emissions.

The recent reform has changed the ‘environmental’ orientation of the VMA by adding ‘allowing sustainable land use’ dimension (see s3(1)(h)). According to the VMFA Bill Explanatory notes (2013:4), this ‘provides for the clearing of native vegetation for sustainable development.’

Similarly to the other statutes, the VMA requires that an entity conferred with a function or power under the Act, performs the function or exercises the power ‘in a way that advances the purpose’ of the Act (s4). To this end, in the SPA-VMA framework, the vegetation clearing regulation is guided by the purpose statement of the VMA and not that of the SPA (i.e. ‘ecological sustainability’).

The regulatory boundaries of the VMA are delineated by two definitions: ‘vegetation’ and ‘clearing’. Section 8 of the VMA defines ‘vegetation’ as:

a native tree or plant other than (a) grass or non-woody herbage, (b) a plant within a grassland regional ecosystem prescribed under a regulation and (c) a mangrove.

‘Clearing’ defines the scope of regulated activities impacting on the state of woody vegetation, which includes (Schedule):

removing, cutting down, ringbarking, pushing over, poisoning, destroying in any way including by burning, flooding or draining’, but excludes ‘destroying standing vegetation by stock, or lopping a tree.
The VMA applies across private tenures, both freehold and Crown leasehold. However, it does not apply to clearing of vegetation on forest reserves, national parks, regional parks regulated under the *Nature Conservation Act 1992* (Qld), an area declared as a state forest or timber reserve under the *Forestry Act 1959* (Qld), or a forest entitlement under the *Land Act 1994* (Qld) (VMA s7). Mangroves as a ‘marine plant’ are protected under the *Fisheries Act 1994* (Qld).

The VMA also provides for formal (statutory) definition of ‘biodiversity’. It defines ‘biodiversity’ as (Schedule):

the variability among living organisms from all sources, including terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part, and includes—

(a) diversity within species and between species; and
(b) diversity of ecosystems.

The SPA-VMA regulation applies only to vegetation clearing activities that are categorised as ‘assessable’ development. In the SPA there is a minimum lot size ‘barrier’ that triggers a ‘material change of use’ and a ‘reconfiguring a lot’ referral to the state (i.e. application of the VMA). The recent reform has modified this ‘barrier’ by increasing the unregulated area from 2 hectares to 5 hectares (SPR Schedule 24, Part 1).

Applications for clearing of remnant vegetation (that is not exempt) can be made only for a ‘relevant purpose’. Section 22A of the VMA specifies a list of ‘relevant purposes’ which include: coordinated projects of state relevance, weed and pest control, public safety, relevant infrastructure activities, fodder harvesting, thinning, clearing of encroachment, and clearing for an extractive industry. The recent amendments to the VMA have considerably expanded the scope of ‘relevant purposes’ by introducing three new purposes (s22A(j), (k), (l)), which are:

- necessary environmental clearing (e.g. restore the ecological condition of the land, natural channel diversion, preparation for natural disaster and decontamination)
- high value agriculture clearing (clearing for annual and perennial horticulture and broadacre cropping)
- irrigated high value agriculture clearing (clearing for annual and perennial horticulture and broadacre cropping, pasture-based farms, which require irrigation).

Agricultural clearing requires the proponent to provide evidence regarding the viability of the proposed development (s22DAB).

**Regulatory instruments**

The VMA framework is built on the regional ecosystem classification system. In Queensland, it was adopted in the 1990s to assist in ‘on and off’ reserve planning for biodiversity. Regional ecosystems have been defined as vegetation communities in a bioregion that are consistently associated with a particular combination of geology, landform and soil (Neldner et al. 2012). Based on this classification and the state of ecosystems, the *Vegetation Management Regulation*
2012 (Qld) incorporates 3 Schedules listing regional ecosystems as: ‘endangered’, ‘of concern’ and ‘least concern’, which then form the basis for protective measures.

The VMA (ss22LA-22LC) prescribes guiding criteria for classifying ‘remnant’ ecosystems based on their pre-clearing extent. Regional ecosystems are:

- **‘endangered’** where the area of remnant vegetation for the regional ecosystem is less than 10 per cent of the pre-clearing extent, or is 10 to 30 per cent of the pre-clearing extent and less than 10 000ha in extent;
- **‘of concern’** where the remnant vegetation extent for that regional ecosystem is 10 to 30 per cent of the pre-clearing extent or, where more than 30 per cent of the pre-clearing extent, the area covered is less than 10 000ha;
- **‘least concern’** regional ecosystems are those where the remnant vegetation is more than 30 per cent of the pre-clearing extent for that regional ecosystem and covers more than 10000ha.

Regulated vegetation management maps are the core instrument that triggers assessment and determines assessment conditions. Under the VMA the ‘regulated vegetation management map’ is the map certified by the Chief Executive ‘as the regulated vegetation management map for a part of the state and showing the vegetation category areas for the part’ (s20A). Maps are prepared and updated by the Queensland Herbarium. Mapping is based on several sources: aerial photographs, satellite imagery, digital elevation models and validated, if possible, through field sampling and ground surveys (Neldner et al. 2012). They are available in digital form.

Property maps of assessable vegetation (PMAVs) (s20AK) provide another important tool of the VMA. The PMAVs can be ordered by landholders to gain greater certainty about which areas can be cleared. They are specifically prepared for the property and identify the status of vegetation at a property scale. Due to greater precision they replace or amend the respective regional ecosystem map for the area (s20AJ). For example, once certified by the Chief Executive, ‘category X’ areas (see below for details) identified on the PMAV can be cleared without a permit.

The VMA prescribes several categories of vegetation maps which are linked to specific assessment provisions. At the end of 2013, the LNP government launched a new mapping system. The VMFAA has introduced new categories of regulatory maps. The major change is the introduction of ‘Category X’ maps, which prescribe all not assessable or not regulated vegetation as ‘exempt’. Category X vegetation can be cleared without permits. As stated in the VMFA Bill Explanatory notes (2013:7):

> these areas will not be ‘clawed back’ over time and be remapped as remnant vegetation, unless the areas are involved in offsetting or subject to restoration/enforcement notices or unlawful clearing. This provides greater certainty for clients when undertaking property planning.

Since the regulations on freehold and indigenous lands restricting clearing of high value regrowth (not cleared since 31 December 1989) have been repealed, ‘Category X’ also can include high value regrowth vegetation.

Activities affecting remnant vegetation for a ‘relevant’ purpose are assessed against criteria of vegetation management codes. Section 11 of the VMA prescribes that the Minister ‘must’ make
codes for vegetation management for regions of the state. The process involves public consultation (s12), and the codes need to be approved under a regulation (s14). The codes also ‘may’ provide for the protection of (s11(2)) (emphasis added):

(a) the habitat of native wildlife prescribed under the Nature Conservation Act as endangered or vulnerable wildlife (protected wildlife); or
(b) a plant that is protected wildlife or least concern wildlife; or
(c) the breeding place of an animal that is protected wildlife or least concern wildlife.

Development assessment codes are also complemented by the offsets policy (s10C).

The recent reform has also introduced the State Development Assessment Provisions (SDAP) (DSDIP 2014e), which set out the matters of interest to the state for development assessment. The SDAP is a compilation of assessment provisions previously contained in different codes. All regional management codes for vegetation clearing are integrated into a single code (Module 8 of the SDAP). All clearing activities are subject to assessment by the Department of State Development, Infrastructure and Planning (DSDIP). Changes in assessment criteria and their implications for vegetation management are detailed in section 6.6.

The reform also introduced self-assessable vegetation clearing codes as another instrument of the VMA. According to the State Policy for Vegetation Management (DNRM 2013c:3):

Self-assessable vegetation clearing codes provide opportunities for landholders to self-manage clearing for low-risk property management activities such as weed control, fodder harvesting, thinning and encroachment; and to improve the operational efficiency of existing agricultural activities, provided they comply with the code and notify the Department of Natural Resources and Mines prior to undertaking clearing activities, where required.

As of May 2014, the DSDIP has published 9 self-assessable vegetation clearing codes.

**Offence provisions**

Another success factor of the SPA-VMA framework was the clearing offence provisions, which were the focus of the previous Labor government. The VMFAA reform has changed several offence and enforcement provisions. It has removed a provision requiring that, in the absence of evidence to the contrary, responsibility for unlawful clearing is placed with the ‘occupier of the land’. It has also expanded defences available to the ‘occupier of the land’ to include ‘mistaken belief’. Amendments have been also made to the Land Act 1994 (Qld) removing the provision that a lease may be forfeited if the lessee has one or more convictions for a vegetation clearing offence.
6.6 Institutional diagnostics: barriers and bridges to biodiversity conservation

The reforms of the SPA-VMA framework have ‘rippled through’ all the major elements of the institutional system. This section presents the findings emerging from the analysis of the SPA-VMA re-design. As the reforms have been introduced only recently, behavioural responses of landholders cannot be fully observed. Therefore, this section examines the system in terms of its potential to ‘maintain protection’ of Queensland’s native vegetation cover.

6.6.1 Regulatory scope and outcomes

<table>
<thead>
<tr>
<th>To what extent:</th>
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<tr>
<td>- does the institutional system provide clear and consistent guidance regarding regulated environmental (vegetation/biodiversity protection) outcomes and their prioritisation;</td>
</tr>
<tr>
<td>- are the environmental outcomes prioritised over competing or conflicting outcomes;</td>
</tr>
<tr>
<td>- does the regulation cover interventions affecting achievement of environmental outcomes (i.e. ‘leaves no gaps’)?</td>
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</table>

Mismatch between regulatory and ecological boundaries is a widely emphasised problem in the environmental management literature. While planning of the protected area network is constrained by tenures and available reserves of state land (see section 5.6), in the SPA-VMA system a ‘mismatch’ of spatial scales does not create barriers for vegetation management.\(^\text{41}\) The vegetation management framework is based on the regional ecosystem mapping which overlays private land tenures and administrative (e.g. local government) boundaries. In this case, of more concern are ‘overlays’ of different state interests, which largely determine the regulatory scope of ‘environmental’ statutes and incorporated gaps.

Compared to the SPA, the VMA pursues a relatively simple purpose, which is the regulation of vegetation clearing to protect a range of environmental values. The recent reform has introduced a conflicting management outcome of ‘sustainable land use’ without specifying priorities. This outcome supports expansion of the scope of ‘relevant purposes’ for vegetation clearing: ‘high value agriculture clearing’, ‘irrigated high value agriculture clearing’ and ‘necessary environmental clearing’. This implies a decrease of limitations placed on activities impacting upon the environmental values, in favour of expansion of intense use agricultural land.

Widening the scope of unregulated clearing on small parcels of land (less than 5 hectares) is hard to quantify. This ‘gap’ may affect some of the wildlife corridors in fragmented landscapes, in particular in the coastal zone (Taylor 2013). Of significant concern, however, are other changes widening the extent of legal vegetation clearing. Their potential ‘on ground’ impacts have been estimated in the recent World Wildlife Fund report (Taylor 2013), which is based on

\(^{41}\) Note: this thesis did not examine the boundary problem in urban areas with small parcels of land.
studies of remnant and remnant regrowth vegetation and good quality agricultural land maps. The summary of the findings is provided in Table 14 (emphasis added).

**Table 14** Estimated impacts of the vegetation management reform, Queensland 2013

<table>
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<tr>
<th>Reform</th>
<th>Estimated (potential) impacts</th>
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<tr>
<td><strong>Removal of restrictions on high value regrowth clearing</strong> (has been cleared previously, not cleared since 31 December 1989) from freehold and indigenous land. Restrictions remain on leasehold land and in 50m protection buffers along certain Great Barrier Reef watercourses</td>
<td><strong>Estimated total area of ‘at-risk regrowth’</strong> is approximately 700,000 hectares. Most at-risk regrowth occurs in the Atherton Tablelands and around Cairns, and in Central and South-eastern Queensland. If the ‘at-risk regrowth’ bushland were allowed to regrow to maturity: - 16 ‘endangered’ ecosystems would revert to ‘of concern’ status - 3 ‘endangered’ ecosystems would revert to ‘least concern’ status - 8 ‘of concern’ ecosystems would revert to ‘least concern’ status</td>
</tr>
<tr>
<td><strong>New clearing purposes:</strong> - necessary environmental clearing - high value agriculture clearing - irrigated high value agriculture clearing</td>
<td>New clearing purpose sets about 1.3 million hectares of mature bushland at-risk in fertile land areas were vegetation has already been significantly cleared. - 10 ‘least concern’ ecosystems could move up to ‘of concern’ status - 8 ‘of concern’ ecosystems could move up to ‘endangered’ status</td>
</tr>
</tbody>
</table>

Source: findings extracted from Taylor (2013)

### 6.6.2 Distribution of roles and responsibilities

To what extent does the institutional system allocate decision-making power, including power in joint decision-making process, to actors who:
- have the necessary capacity to undertake the allocated scope of actions;
- have a commitment to protect environmental values?

To what extent does the institutional system enable the stakeholders (public) to influence/challenge decisions of the authorities?

Development assessment outcomes are affected by the interests of involved actors. Environmental outcomes will broadly depend on two factors: first, interests and capacities of the regulatory authority to constrain the rights of developers; and second, interests of developers to protect environmental values. Of particular importance these factors are in performance-based systems, which allow for considerable scope of discretion and interpretation in determining land use.

At the regulatory level, the threat to the protection of environmental interests is that authority over design of integrated planning instruments (e.g. State Planning Policy) has been allocated to the Department of State Development, Infrastructure and Planning (DSDIP). The DSDIP pursues a vision to ‘drive the economic development of Queensland’, supported by such objectives as ‘champion the interests of business and industry in Queensland’ and ‘fast track delivery of major resource and industrial development projects’ (see DSDIP 2013a). The changes in regulatory instruments introduced by the DSDIP are described in 6.6.3.

Further integration of development assessment process has led to the creation of a new centralised State Development Assessment Agency (SARA), which controls and coordinates incorporation of state interests in the development assessment. The SARA sits within the
DSDIP (see Table 10). This organisational arrangement may significantly modify and reduce the effectiveness of environmental restrictions and conditions incorporated in the assessment to mitigate adverse development effects.

Another concern is the considerable increase in self-assessable development. The recent introduction of 9 self-assessable development assessment codes indicates that many assessment decisions will be made by landholders. In light of the incentives to pursue land clearing (see section 6.6.5), ‘self-assessable’ clearing for encroachment, fodder harvesting, thinning and weed management, may result in deliberately or mistakenly overlooked, or misinterpreted, requirements.

Another important protective factor in value conflict situations is the right of the public to challenge decisions of the regulatory authority. The VMFAA reform has removed previously allocated appeal rights against decisions made by the Chief Executive, which includes certification of regulatory maps. This places the decision-making power entirely in the hands of the Chief Executive, and this allows, if politically required, supporting a pro-development direction instead of protecting environmental values.

6.6.3 Decision-making instruments

<table>
<thead>
<tr>
<th>To what extent:</th>
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<tr>
<td>- do the regulatory instruments require or enable the regulatory authority to protect ecological systems or their properties (e.g. prohibit or constrain development);</td>
</tr>
<tr>
<td>- does the allocated scope of authority requires or enables the regulator to respond to observed or predicted changes in the state of the environment?</td>
</tr>
</tbody>
</table>

The success of the SPA-VMA framework in achieving a decline in vegetation loss is linked to the design and operation of several regulatory instruments. As a statutory system, the SPA-VMA is reactive in nature. Therefore, the core ‘environmental performance’ dimensions are the rules which enable authorised agencies to prohibit/constrain development (i.e. vegetation clearing) in ecologically significant areas or minimise/mitigate adverse effects.

The VMFAA reforms have weakened the protective measures of many instruments of the SPA-VMA framework. Introduced as part of ‘streamlining’ and ‘integration’ agenda, less visible changes concern the content of codes prescribing assessment criteria for vegetation clearing. Table 15, which contains a summary of one of the published reviews, illustrates the nature of the strategy employed by the regulator (emphasis added).

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42 Note: these rights now are limited to those decisions affected by jurisdictional error.
### Table 15 Changes in vegetation clearing assessment requirements

<table>
<thead>
<tr>
<th>Vegetation clearing</th>
<th>Offsets</th>
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</thead>
<tbody>
<tr>
<td><strong>Acceptable outcomes for wetlands are relaxed</strong>: allowing the clearing of vegetation (at specified widths and with clearing limits) within 50 metres of the edge of the wetland in prescribed circumstances;</td>
<td><strong>Expanded definition</strong> of “maintaining the current extent” allows the applicant to propose a land-based offset to satisfy a required outcome without demonstrating that avoidance was not reasonable practical;</td>
</tr>
<tr>
<td><strong>Acceptable outcomes for waterways are relaxed</strong>, with clearing generally allowed to within 10 metres of Stream Order 1 and 2 watercourses at specified widths;</td>
<td><strong>Allows the offset area to be smaller than the impact area if an Ecological Equivalence Assessment finds that the values for the offset area significantly exceed the clearing area;</strong></td>
</tr>
<tr>
<td><strong>Offsets are permitted as an acceptable means</strong> of meeting performance objectives for connectivity in most circumstances (previously some regional codes specifically excluded offsets);</td>
<td><strong>Allows offsets on slopes of up to 20% and within wetland protection areas;</strong></td>
</tr>
<tr>
<td>The <strong>requirement to maintain connectivity</strong> to mapped remnant native vegetation on adjacent properties has been removed for extractive industry and high value agriculture clearing:</td>
<td><strong>Deletes the requirement to provide offsets for wetlands that have the same or higher wetland status than the wetland being cleared;</strong></td>
</tr>
<tr>
<td>Clearing of Essential Habitat is permitted in most circumstances provided that clearing does not exceed specified widths and areas; and,</td>
<td><strong>Deletes performance requirements for Threshold Regional Ecosystems.</strong></td>
</tr>
<tr>
<td>In many circumstances, <strong>clearing of ‘endangered’ and ‘of concern’ regional ecosystems is permitted</strong> providing that it occurs at specified widths which respect designated buffers.</td>
<td></td>
</tr>
</tbody>
</table>
6.6.4 Environmental information

To what extent can the actors in regulatory positions obtain information on past, existing or potential environmental pressures (including cumulative)?
To what extent does the prescribed environmental information enable the actors to understand the situation (e.g. assess impacts, predict the response to interventions etc.)?

The planning and development assessment system requires a significant level of environmental knowledge. This study could not assess ecological ‘soundness’ of the remnant vegetation mapping system underpinning the SPA-VMA framework or the quality of information available to decision-makers (see section 5.2.4 for limitations). However, several changes in the regulatory system may influence further developments of the information base.

Along with environmental impact assessments, site surveys and other information requests are important tools in decision-making process. To this end, one of the information barriers for the regulators is that of data gathering costs. In the SPA-VMA context, the nature of the ‘green-tape’ reduction strategies suggests that environmental information requirements will be lowered to speed up the development assessment process, and to decrease market entry costs and time for developers.

It is acknowledged, that despite technology advances, site-specific biodiversity data is incomplete (Australian Government 2009). Remnant vegetation maps are based on satellite imagery which is not precise. Flexible provisions combined with PMAVs (see section 6.4.2.) allowed making corrections in response to new information. Introduction of Category X maps suggests that the government will experience a decrease in the accessibility of site-specific environmental information.

6.6.5 Incentives and disincentives

To what extent do the rules:
- provide for incentives for landholders to prioritise environmental values or minimise trade-offs;
- reduce incentives for landholders to adversely affect ecological systems or their properties?

To what extent do the rules provide incentives for regulators to prioritise environmental values or minimise trade-offs leading to permanent loss or destruction of environmental values, including biodiversity?

Changes in regulation as such do not produce environmental impacts. In the SPA-VMA context, the scope of ‘on ground’ environmental impacts will largely depend on the rules determining motivation of the regulator to perform its role and prioritise environmental values, and those affecting agricultural landholders and their interests in the use of allocated clearing rights.

Examination of interests/capacities/knowledge of individual landholders has not been the focus of this study. However, the analysis of dominating drivers of biodiversity loss, such as
agricultural export markets (see section 5.3.1), suggests that landholders would not undertake the burden of vegetation/biodiversity conservation at the expense of their income. It is generally known, that choices of agricultural producers are influenced by another powerful ‘regulatory system’, which is banking. This system can also influence (e.g. via mortgage agreements) their abilities to conclude conservation agreements or undertake carbon farming. Generally, cleared agricultural land has a higher market value (Dupont 2014).

In this context, there are two major concerns. First, it is highly likely that the new ‘regulatory gaps’ (see section 6.6.1 for spatial extent) will result in significant increase in remnant vegetation clearing undertaken by landholders. Second, the rate of illegal clearing, as well as ‘self-assessable’ clearing, which break the requirements of the codes, may increase in light of changes in offence provisions. The overall rate of illegal clearing, however, may decrease due to increase in areas available for clearing, which legalises previously prohibited activities.

As identified in chapter 5, a recent trend is the declining interest of the Queensland government in protection of environmental values. The changes in the SPA-VMA framework reflect the changes in political values introduced by the LNP government focusing on an economic growth agenda. Political preferences towards agricultural development are clearly expressed in statements by the Minister for the Queensland Department of Natural Resources and Mines (DNRM), made during the reform period (QCMD 2013):

> The Government has made it clear it plans to double the value of agricultural production in Queensland by 2040, and these reforms are a vital component to achieving that goal.

> The previous Labor government had allowed the pendulum to swing too far towards extreme green policies, threatening the ability of landholders to effectively manage their businesses and maintain productivity.

> These new laws strike a balance between agricultural production and environmental protection, and represent the most significant reforms to legislation affecting the rural sector in decades.

This political direction suggests that the regulator will continue to increase access to productive land resources at the expense of environmental values. This may also suggest that low priority will be given to environmental performance monitoring and sanctioning activities.

The SPA-VMA framework does not prohibit local governments to undertake the role of environmental regulator. However, shifts in land use, which negatively affect the market value of land (e.g. de zoning), involve the issue of compensation claims. This problem is not limited to biodiversity issues. As identified by the Productivity Commission (2012), these disincentives restrict local governments in applying measures to prevent development in areas subject to environmental hazards such as sea level rise.

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43 Illegal clearing is still a problem in Queensland. For example, in 2006 the SLATS satellite imagery system identified 61000 hectares of potential illegal clearing in Queensland (DOE 2006).

44 Note: DNRM is responsible agency for the implementation of the VMA (see Table 10).
6.7 Summary and conclusions

This chapter completes the second part of this study examining institutional drivers of biodiversity loss in the state of Queensland, Australia. It examined the environmental performance of the Queensland Planning System by focusing on the regulatory framework of the Sustainable Planning Act 2009 (Qld) (SPA). It also examined a specific development problem - vegetation clearing - which is the major cause of terrestrial biodiversity loss in the state. The scope of the analysis incorporated provisions of the Vegetation Management Act 1999 (Qld) (VMA), which form part of the development assessment system of the SPA. The SPA-VMA framework is the major institutional determinant of biodiversity protection outcomes that can be achieved outside specifically designated areas.

In the past, the VMA-SPA framework has been linked to success in decreasing vegetation clearing rate at the state scale. In 2013 the LNP government undertook a major reform of the framework, which sought to advance its vision of prosperity by doubling agricultural production by 2040. Therefore, this chapter aimed to examine how the reform has affected the potential of the SPA-VMA framework to protect the native environment from vegetation clearing pressures, and to reduce the rate of biodiversity loss.

As the study reveals, the recent reform has significantly changed the regulatory structure of the framework across all performance dimensions examined. On the one hand, the reform has clarified several elements of the system and reduced complexity of the framework through the integration and streamlining of regulatory provisions. On the other hand, it has: widened regulatory gaps, increased possibilities for broadscale vegetation clearing for agricultural purposes, contributed to the loss of valuable remnant vegetation, weakened ‘protective’ strength of the instruments, excluded the possibility of regulatory response to new environmental information, allocated decision-making control to the regulator pursuing a development agenda, and removed the rights of the public to challenge its decisions. The reform has also shifted a significant part of assessment responsibilities to landholders, and has weakened offence provisions.

In conclusion, it remains to be seen whether under these ‘rules of the game’ the SPA-VMA framework will be able to achieve its reform goal of ‘protecting’ native environment in the light of increasing agricultural pressures. The reforms are recent and the behavioural responses will become evident only with future monitoring of actual vegetation clearing. However, it is highly likely that vegetation clearing in Queensland will increase to higher levels thus leading to further loss of terrestrial biodiversity.
Chapter 7 - Synthesis, Reflections and Concluding Remarks

7.1. Introduction

This study was guided by an interest in the problem of loss of terrestrial biodiversity caused by human actions in the state of Queensland, Australia. The problem was causally linked to the system of rules regulating land use planning and development assessment decisions. The study started with a predefined question of how effective is this institutional system in preventing biodiversity loss. The search for a theoretical and analytical platform for effectiveness evaluation significantly expanded the scope of this study. As a result, the study explored two major research questions:

- How to conceptualise and evaluate the effectiveness of institutions contributing to the resolution of environmental problems?
- How effective is the Queensland land use planning and development assessment system in achieving biodiversity protection outcomes?

The study evolved in an iterative fashion. Many sub-questions emerged from the necessity to address gaps that were encountered in the literature, or to clarify theoretical, conceptual or analytical questions. For the sake of clarity and logic of information flow, this iterative process was not made explicit in the previous chapters. Therefore the major aim of this concluding chapter is to reflect on the evolution of this study, outline its major contributions and findings in relation to the research questions, and to identify research gaps and opportunities for further studies.

Following the structure of this thesis this chapter consists of three major parts. The first (section 7.2) addresses the theoretical and analytical problems detailed in chapters 2, 3 and 4. The second (section 7.3) reflects on the evaluation process of the Queensland Planning System and its outcomes detailed in chapters 5 and 6. The third (section 7.4) identifies major limitations of the study and future research directions.

7.2 Institutions, effectiveness and biodiversity protection problem

7.2.1 Institutions, institutional functions and institutional effectiveness

Chapter 2 was devoted to a range of theoretical, conceptual and analytical questions about the conceptualisation of institutions, their causal role and evaluation. More specifically, it addressed four major questions:

- What is understood by ‘institutions’?
- What role do institutions play in structuring the behaviour of human actors?
- How can the effectiveness of institutional performance and institutional environmental performance be conceptualised?
- What are the core elements of an assessment design?

Its aim was to establish the theoretical and analytical platform for this study. However, many of the findings, as well as the proposed theoretical and conceptual solutions, may also have a wider applicability for further institutional studies.

**Institutions and institutional functions**

Clarification of the core concepts is an essential point for any study. This study commenced with a discussion of the concept of ‘institutions’. It was based on the theoretical platform of New Institutionalism, which largely draws on a common understanding of institutions as ‘systems of rules’ or, as Rational Choice institutionalists (e.g. North 1990, Ostrom 2005) emphasise, ‘rules of the game’. New Institutionalism provides a broad theoretical umbrella for various disciplines examining the institutional role in social interactions.

Various schools of thought offer different definitions. The study did not attempt to engage in the theoretical debate regarding ‘how far we can stretch the meaning of the term rule’ (Hodgson 2006:4) in a definition. Instead, it accepted the broad understanding of ‘rules’ as shared social constructs that structure human interactions. This study made a distinction between two major types of institutions: ‘formal’ (written) and ‘informal’ (unwritten). It also accepted the distinction between ‘institutions’ and ‘organisations’ as composite institutional actors, and ‘institutional’ and ‘organisational’ rules. This approach allowed consideration of a broad scope of rules, and offered a good ‘conceptual roadmap’ enabling navigation of different application contexts of the term, and of distilling findings relevant for the study.

The analysis of institutional effectiveness is inevitably linked to the question of how to distinguish between institutional and non-institutional factors influencing human behaviour. The question regarding the causal role of institutions turned out to be a significant challenge for the research process. New Institutionalism covers a range of disciplinary perspectives on institution-actor relations, which tend to explain specific behavioural phenomena, social practices or modes of interactions. This diversity of theoretical insights has produced a rather confusing picture of institutional determinants of human interactions.

The work on the theoretical platform was not initially included in the scope of this study. The need to reconcile different perspectives and to create some common theoretical ground emerged out of two problems identified during its course. First, significant confusion and inconsistency was observed in academic discussions regarding institutional determinants of environmental problems. Second, a detailed review of the biodiversity protection problem made it clear that, on the operational level, the institutional role cannot be reduced to a limited set of functions, such as incentive-sanction mechanisms, decision-rules, or rules contributing to the construction of roles or identities. In other words, in approaching institutions from the problem perspective this study, it could not be ‘grounded’ in any single theoretical framework established under the New Institutionalism umbrella.
Based on extensive reading of the theoretical literature, particularly the studies by North (1990), Sharpf (1997), Peters (1998), Scott (2001, 2008), March and Olsen (1989, 2008) and Ostrom (2005), the study developed a new typology of institutional functions. This was done by considering the different ways in which the rules can affect the three behavioural properties of human actors: perception, preferences and capabilities. As a result, eight generic institutional functions were proposed: coordinative, constitutive, cognitive, normative, payoff, constraining-enabling, allocating and structuring. While this typology might not be useful for all cases, the study argued that it captures the diversity of institutional effects distinguished in the theoretical and empirical literature, and demonstrates that institutions are important explanatory factors for a large variety of behavioural phenomena.

**Institutional performance and effectiveness**

Chapter 2 also addressed several conceptual and analytical problems related to the evaluation of institutional performance and institutional environmental performance. Initially, the description of assessment design was planned as a part of the methodology section. However, the review of the evaluation literature revealed that the concepts of institutional performance and institutional effectiveness are poorly defined, and significant confusion exists regarding the core elements of assessment design.

The development of the conceptual and analytical platform for this study was significantly assisted by the Institutional Dimensions of Global Environmental Change (IDGEC) research project literature (Young 2002, Miles et al. 2002, Underdal & Young 2004, Young et al. 2008, Breitmeier et al. 2011). This project was devoted to a systematic and comprehensive analysis of international environmental regimes. At the same time, it detailed many conceptual and analytical problems encountered in institutional analysis. Building on the conceptual and methodological platform of this project, this study further clarified the core elements of assessment design and made the necessary adjustments to support the analysis of other types of institutions.

The development of the conceptual framework was driven by the definitions of two major concepts proposed by this study: ‘institutional performance’ and ‘institutional effectiveness’. As the rules are only social constructs they cannot ‘perform’ directly. Therefore, the framework was built on the premise that institutional performance can materialise only in the form of the behavioural response of regulated actors. Therefore, ‘institutional performance’ was defined as an institutional influence on, or contribution to, the behavioural response of targeted actors. Consequently, the concept of ‘institutional effectiveness’ was defined in terms of the extent to which an institution influences, or can potentially influence, the behavioural response of targeted actors towards the outcomes an institution has been designed to produce. Institutional ‘environmental’ performance was further linked to the behavioural response of targeted actors,
which, either directly or indirectly, produces or has a potential to produce, environmental effects (impacts).

This definition of institutional performance highlighted another question: what exactly is evaluated? To define the assessment object, the study proposed another concept so far not explicitly identified in the evaluation literature: the ‘institutional performance dimension’. The rationale for the concept was the complexity of institutional designs that precludes analysts from an evaluation of the whole set of rules. Therefore, the explicit recognition of a ‘performance dimension’ as an assessment object makes a clear distinction between institutions as complex systems consisting of different clusters of rules, and the groups of rules being evaluated. Conceptually, ‘performance dimensions’ were distinguished from a large variety of criteria that could be selected for their assessment.

The evaluation literature offers different approaches to assessment design. However, the concepts of ‘effectiveness’ and ‘performance’ tend to reflect a managerial or result-based perspective, implying availability of observable results, clear goals, performance standards and measurable evaluation criteria. In practice, as the IDGEC project literature demonstrates, the attempts to conduct the assessment along these lines encounter numerous methodological challenges. These include vague, changing or no-endpoint goals, lack of performance standards, incomplete data and, most importantly, the problem of complex causality. Similar problems were also encountered in the assessment of the Queensland Planning System (see section 7.3).

Based on the literature review, the chapter 2 was summarised several challenges encountered in the evaluation of institutional environmental performance. These problems underpinned the core argument made by this study, that despite the interest in ‘tangible results’ characterising the nature and extent of behavioural change, most institutions would require performance dimension- or rules- based evaluation. Consequently, this study proposed to conceptualise the effectiveness of institutional environmental performance, both in terms of the institutional contribution, and of the institutional potential to contribute to the required change in the behaviour of regulated actors.

Another major output of chapter 2 was a conceptual framework clarifying core concepts and outlining the different approaches that could be taken to institutional performance assessment. This framework allowed the linking of findings from various evaluation studies, as well as the examination of different evaluation designs and their outcomes.

7.2.2 Biodiversity protection and institutions

Chapter 3 examined the scope and nature of the biodiversity protection problem, which is the focal environmental problem selected for this study. The major aim of this chapter was to identify institutional determinants of the problem, and to characterise criteria which could
underpin the analysis of institutional environmental performance. The review was guided by two broad questions:

- What role do institutions play in structuring human interactions with biodiversity?
- What performance dimensions and criteria can characterise an institutional potential to achieve biodiversity protection outcomes?

The search for institutional design features that contribute to the achievement of sustainable use of environmental resources and biodiversity has formed a significant part of institutional studies. The major focus so far has been on institutional mechanisms shaping the motivation of involved actors to conserve environmental resources or perform environmental duties. This study aimed to examine a broader range of functions that institutions could potentially perform in structuring human-environment interactions.

The starting point for the review and also for this study was the problem of science-policy integration, or incorporation of discovered ‘laws of nature’, into the rules structuring human-environment interactions. However, a large variety of problem attributes identified in the literature led to the gradual expansion of the review. Consequently, using a set of generic questions - what should be protected, where, why, how and by whom - the chapter examined the various attributes of the biodiversity protection problem.

Institutional designs have been acknowledged as a barrier to sustainable management of ecological systems and biodiversity. However, the review revealed that academic literature is plagued by three major problems. First is the fragmentation of knowledge. While numerous studies consider different attributes of the biodiversity problem, the literature has been fragmented into so many disciplinary specializations that it is hard to link and integrate findings. Second is the conceptualisation of institutions. A lack of conceptual agreement, particularly in the environmental sciences cluster, has created significant confusion about institutional and non-institutional determinants of environmental problems. This has made a systematic literature review highly difficult. Finally, the third problem lies in the vague and generic statements made in the literature. Recommendations tend to be stated in the form of desired outcomes, such as ‘the need to manage adaptively’, ‘integrate environmental considerations’, or in a few cases, even more simply to ‘remove institutional barriers’. What actions are hindered or enabled, what institutional mechanisms are at play or create so called ‘barriers’, or how to redesign the institutions, has been rarely addressed in detail.

Therefore, the major contribution of this study was an attempt to re-assemble the different pieces of the biodiversity protection ‘puzzle’ into a more holistic and comprehensive picture. Chapter 3 sought to distil a set of problem attributes and their institutional determinants from different streams of the literature which, taking into account different disciplinary backgrounds and theoretical frameworks, are rarely explored together. Using a set of generic questions allowed distinguishing between the ‘science’, ‘management’ and ‘governance’ clusters, which
offered different yet complementary insights into various dimensions of the biodiversity protection problem.

As the literature review revealed, biodiversity protection is not just the maintenance of species richness or the conservation of pristine environments. Instead, it is a highly complex and multi-faceted problem. Institutional designs need to address a range of problem attributes, such as the existing knowledge base, value and incentive systems, distribution of decision-making authorities, and coordination of interactions between multiple actors. Thus, the full spectrum of institutional functions could be ‘at play’ and contribute to the resolution or creation of the ‘on ground’ environmental problems, and all should be dealt with in institutional analysis.

The study could not provide a definitive answer to the question of what criteria can characterise an institutional potential to achieve biodiversity protection outcomes. Different disciplines have provided a large variety of insights on what interventions ameliorate, or cause the problem of, biodiversity loss. Among the dominant themes are changes in resource management theories, involvement of scientists and knowledge holders in decision- and policy-making processes, valuation of ecosystem services and biodiversity, design of flexible (adaptive) management systems, cross-sectoral cooperation and coordination, devolution or redistribution of regulatory roles and responsibilities, design of new collaborative management bodies, as well as the introduction of private property regimes and market mechanisms.

From an evaluation perspective the diversity of propositions and findings raised the question of how to approach proposed solutions. As observed in other review studies (e.g. Lemos & Agrawal 2006), there are two major streams in the literature. The first contains the work of scholars driven by the search for some ideal solution to particular problem attributes. In these studies proposed solutions, such as adaptive management, collaborative approaches, and community involvement, have been presented in a normative fashion as something basically good or desirable. In contrast, the other stream, which also includes the IDGEC project analysts, points to the complexity of social and ecological systems and suggests caution regarding ‘blueprint’ solutions. These studies emphasise the need to carefully examine the assumptions underpinning a particular solution, and the variety of conditions that could contribute to the success or failure of implementation.

These two streams explain the two different evaluation strategies applied in academic literature. The first provides the rationale for an analytical approach which builds on designing some ideal template, such as deriving a set of (ecosystem) management or governance principles that should be considered in institutional designs. The second excludes the possibility of identifying a required set of conditions for success (i.e. effectiveness criteria) across problems and contexts. Instead, drawing an analogy with medicine, the analysts suggest a ‘diagnostics’ approach. It implies examining how institutional designs address a range of problem attributes relevant for the regulation in each particular context. In the literature this conceptualisation of institutional
effectiveness has been labelled as a ‘problem of fit’ (Goodin 1998a, Young 2002, Galaz et al. 2008).

The starting point for this study was the search for a set of value criteria for institutional assessment. The initial research strategy attempted to distil solutions, which, based on some empirical studies, or on general agreement among the scholars, were positively linked to the resolution of the biodiversity problem. This strategy was abandoned following closer examination of the concept of adaptive management which, despite its popularity, has failed to provide a common decision-making structure. As a result, the study shifted to the ‘diagnostics’ path, following the premise that there is no universal regulatory solution to any attribute of the biodiversity protection problem. Therefore, a different ‘logic’ was applied to the literature review, which then involved learning from the debate and from both successful and unsuccessful cases. None of the solutions has been approached from the position of ultimate success or failure. The review, and the problem aspects it identified, set the foundation for the development of the diagnostic framework.

7.2.3 Institutional diagnostics

Chapter 4 concluded the first part of the study by proposing a structured approach to the evaluation of institutional environmental performance. Taking into account the complexity of institutional systems and diversity of their designs the chapter focused on two analytical questions:

- How to examine diverse determinants of institutional environmental performance in a systematic way?
- How to perform the analysis of diverse institutional designs?

The review of the biodiversity protection problem raised the question of how to approach the large diversity of problem attributes distinguished in the literature, and how to specify the implications they have for institutional design and performance. In response, the study examined an application of ‘institutional diagnostics’ which is another major contribution of the IDGEC research project (Young 2002, 2008). Institutional diagnostics is a systematic process for identifying and structuring different attributes of environmental problems, and examining their implications for institutional design. The approach has been described in the context of new institutional designs (Young 2002). In turn this study extended its application to the development of a framework for the evaluation, or ‘diagnostics’, of established institutional systems.

The diversity of institutional designs highlighted another question: how to disaggregate these complex systems into a set of design features (performance dimensions), and to diagnose their actual potential to address identified problem attributes. Recognising the fragmentation of knowledge and the problems with linking different findings, the aim of this study was to find and to apply an existing framework shared by many analysts. The literature revealed (Blomquist
& de Leon 2011) that so far one of the most widely applied and empirically tested frameworks has been the Institutional Analysis and Development (IAD) framework. Drawing on the game-theoretic approach, the IAD framework identifies a group of elements that structure decision-making situations (‘arenas’). These elements are then linked to seven generic groups of rules.

Chapter 4 demonstrated how ‘institutional diagnostics’ and the IAD framework can provide a sound foundation for the design of a framework for institutional analysis. The development process was described as a set of three interacting steps or procedures. Upon reflection, the major discovery made during the design process in this study was that a large diversity of institutional variables, addressed in the environmental science, ecosystem management and environmental governance literature, can be organised building on the typology of rules of the IAD framework. This suggests that, while the game - theoretic approach has been regarded as an analytical tool of economists, ‘the logic of the game’ also offers a good platform for analyses of different types of interactions and their institutional settings.

The main output and a major contribution of this study to the analysis of institutional environmental performance is the design of a new diagnostic framework. It covers the two hierarchical levels of rules most directly affecting ‘on ground’ environmental outcomes and resulting impacts. The framework also links the diverse determinants of institutional environmental performance, presented in the academic and political literature, into one coherent structure. The purpose of the framework is to provide a diagnostic tool, which can assist analysts in selecting, mapping and analysing configurations of rules that can potentially operate as determinants of environmental outcomes in specified settings. To this end, it can be regarded as a bridge between the studies analysing different aspects of environmental problems, and those examining institutional designs.

### 7.3 Queensland Planning System: evaluation and outcomes

Chapters 5 and 6 addressed the question of the effectiveness of the Queensland Planning System in achieving biodiversity protection outcomes. The evaluation was designed as a desktop study based on documentary sources. Data scoping and analysis used the structure of the diagnostic framework in identifying the most important clusters of rules structuring the ‘biodiversity protection game’. The analysis was guided by several groups of diagnostic questions adapted from the diagnostic framework developed in chapter 4.

The analysis was approached from two perspectives. From the broader perspective in chapter 5, it was viewed as a complex interacting system of rules, which cumulatively determine the current land use pattern in Queensland, and the interests and capacities of environmental regulators. From the narrower perspective in chapter 6, it examined the regulatory framework established under the Sustainable Planning Act 2009 (Qld) (SPA). The SPA is an important regulatory sub-system which integrates several statutory frameworks through the development
assessment process. As a result, it provides for the regulation of a broad scope of development activities impacting on the environment, including vegetation clearing.

Several findings can be drawn from both the research process and the outcomes. The first part of this section reflects on the structure of the study, application of the diagnostic framework and data scoping and analysis. The second part describes the major findings derived from the evaluation of the Queensland Planning System.

7.3.1 Reflection on research design and analysis

The initial scope of the study was limited to an analysis of the regulatory framework established under the SPA. The analysis of the land use planning schemes, which are a major regulatory instrument of the SPA, identified a set of barriers limiting consideration of biodiversity issues. However, it failed to provide an answer about the options the regulators have in their choices of rules and the constraining factors. As a result, the study was extended to incorporate a broader system of rules regulating the use of land and environmental resources at both the state and national level. This also corresponded to the two levels of rules recommended in the application of the IAD framework (Ostrom 2005).

Applying the proposed diagnostic framework for data scoping and analysis had both benefits and challenges. As biodiversity loss is an obvious problem, it is relatively easy to conclude that the entire institutional response is ineffective. However, a far more complex problem was to establish what barriers exist within the system that preclude an effective response, and to make predictions about future directions. In this context, the major benefit of the diagnostic framework was that it facilitated an understanding of the complex, dynamic and ‘patchy’ systems of rules which organise interactions in a ‘real life’ government setting. The ‘logic of the game’ allowed dealing with the complexity of the system, and distinguishing different sub-systems of rules that are most likely contributing to the problem, and examining their interactions.

The major challenge was the complexity of the analysis. Applying the diagnostic framework required not only establishing causal relations between a particular environmental problem and its institutional drivers, but also identifying critical linkages between the sub-components of the whole institutional system. In the case of the Queensland Planning System, measures creating barriers, or contributing to biodiversity protection outcomes, can be contained in both national and state strategies, laws and regulations, the common law system, as well as policy instruments developed at the local government level. As a result, the whole data gathering process resembled systems modelling by trying to identify crucial components from the complex network of rules which define the nature of the regulatory response. In diagnostics of such complex systems missing links and some misinterpretation are almost unavoidable.
Several findings were made during the data scoping process. In chapter 5 this study took a broad approach to the scoping of rules without restricting them to particular categories such as statutes, strategies or policies, or themes such as ‘environmental’ regulation. This approach was not intended in the initial scope, but evolved during the process of document analysis.

Environmental governance is not an isolated system. The causal chain to ‘on ground’ biodiversity loss is quite long, and includes rules which cannot be qualified as ‘environmental’, such as GDP/GSP reporting, taxation system, fiscal strategies and state budget. At the same time, they drive regulatory responses and determine their nature and priorities. Similarly, each system is built on deeply embedded ‘fundamental’ rules, such as fundamental legislative principles that largely determine the possible scope of actions and regulatory responses. Omitting these rules from the analysis, or ignoring their influence, may significantly affect the quality of the analysis and the credibility of findings and recommendations.

Another finding, which supports statements made by other analysts (see section 2.3.1), is the lack of fixed and clear goals, performance standards and indicators. In a real life setting, biodiversity protection can be described as an ongoing, dynamic problem, requiring invention of different regulatory responses to changing environmental, social and economic conditions. ‘To stop the decline’ in biodiversity is a high-level target (NRMMC 2010). However, there are no commonly accepted standards describing the state of the environment which would indicate that the problem is resolved, i.e. ‘decline is stopped’. Measurable targets, such as the desired spatial extent of protected areas, tend to be established, but they are not observed and fluctuate with changes in policy direction.

In practice, the effectiveness analysis of dynamic institutional responses to such complex problems cannot be designed using the methodological guidelines developed for the analysis of small scale programs or policies. As was experienced in the research process, pre-structuring the study based on boundaries of a single regulatory framework or instrument not only limits the analysis, but also affects the quality of findings. Gaps in methodology for the analysis of complex institutional systems will be outlined in section 7.4.

7.3.2 Queensland Planning System: can it protect biodiversity

Biodiversity decline is an ongoing problem in Australia and in Queensland. It is causally linked to extensive use of land resources. The state of Queensland has a resource based economy. More than 90 per cent of its land is allocated for development activities. The dominant land use is agricultural and pastoral production, which occupies approximately 80 per cent of land. The land classified as residential accounts for only 1 per cent of the total area. However, it is concentrated in a narrow coastal zone. The mining sector covers a limited area (0.4 per cent), but is going to expand with the growing global demand for mineral and petroleum resources.
Biodiversity loss is largely the legacy of past land use decisions made over 200 years of European settlement history in Australia. However, vegetation clearing for agricultural use, expansion of residential and industrial areas along the coastal zone, and grazing pressures are an ongoing threat contributing to the further loss of native ecosystems. Land degradation has flow on effects on the quality of freshwater and marine ecosystems.

Australia is a contracting party to many international agreements designed to protect the environment, including the Biodiversity Convention. The distribution of regulatory powers among the three tiers of government is set in the Australian Constitution. The regulatory role of the federal government in environmental matters is largely determined by international obligations. The federal government is also responsible for the taxation system and collects and distributes the largest part of taxation revenues. Since the 1990s the major role taken by the federal government in biodiversity protection matters is that of initiator, coordinator and co-funder of national environmental policies. Some of these policies have set the scene for changes in statutory and strategic frameworks in the states and territories.

The state and territory governments are the major regulators of land use planning, resource extraction and other development activities. They are also the owners (on behalf of the Crown) of land resources reserved for public purposes, including biodiversity conservation. The interest of the Queensland government in biodiversity protection matters is a crucial determinant of the regulatory responses and their outcomes. Local governments play a minor role in environmental regulation. However, they have the responsibility for land use planning schemes, which are a central instrument for regulating land development activities.

Over the past three decades there has been a rapid expansion of the regulatory system in Australia and in Queensland, which is directed towards achieving sustainable use of biological resources and protection of environmental values. Approximately 5.1 per cent of Queensland’s land is allocated for nature conservation and protected under national parks tenures. The network of private protected areas is also expanding. Protection regimes have been also established for tidal vegetation and threatened wildlife. Significant achievement has been an introduction of the politically difficult *Vegetation Management Act 1999 (Qld)* (VMA), which limits the clearing of remnant vegetation on freehold and leasehold tenures. New incentive-based instruments are also evolving.

The analysis of regulatory responses also suggests that significant problems must be faced if Queensland is going to ‘halt’ biodiversity decline. As state of environment reports identify (e.g. DEHP 2012), the extent of protected areas is still insufficient to protect diverse ecological systems. Regulatory responses have reduced, but not ‘halted’ vegetation clearing. Cattle grazing pressures remain largely unregulated and fluctuate in response to climatic conditions and global market demands. Mining industries compete for land resources and are given priority in nature refuge areas protected on private land.
The recent shift in political direction in the Queensland government set a new trajectory for regulation. Focus on agricultural and resource pillars of the economy, regulatory cost reduction, and ‘fast-tracking’ of development have led to changes in the regulatory system. The LNP government enabled further clearing of remnant vegetation for intensive agricultural use purposes, enabled mining activities in some protected wild river areas, and introduced ‘emergency drought’ grazing practice in some national parks. The government has weakened the protective strengths of a range of other regulatory instruments in favour of agricultural development and resource extraction. Overall ‘on ground’ effects of regulatory changes are hard to assess, as they depend on the response of regulated industries and other institutional drivers such as global markets. However, the increase in vegetation clearing rate and loss of remnant vegetation in agricultural areas is likely to continue.

The SPA-VMA framework is a central regulatory system which places constraints on vegetation clearing activities. Detailed examination of the system, conducted in chapter 6, has revealed further gaps in the vegetation management system. It allows the clearing of endangered native ecosystems and habitats of threatened species. These gaps have been ‘widened’ through the recent (2013) regulatory reform, which has removed and relaxed many of the previously established constraints.

Australia is a developed country and has one of the largest world economies. Many problems with regulatory responses to biodiversity protection, can be traced back to the fundamental question: why protect? In Queensland land is seen as a resource supporting economic growth. Under the utilitarian approach, which dominates the current political environment, ‘biodiversity’ has a value as long as it can contribute to growth. In this context, protected areas are seen as ‘recreational’ service providers and are required to boost the tourism industry.

Biodiversity does not yet have a stable market, nor social value. Therefore, balancing this interest against ‘prosperity’ goals, measured in terms of Gross State Product is difficult. Moreover, conservation measures form an ‘expense position’ in the state and local government budgets, which currently struggle to find the balance between unpopular taxation measures and growing public demand for services. Therefore, it can be expected that the interest of regulators in development and application of biodiversity protection measures will decrease.

In summary, the Queensland Planning System incorporates many potentially effective regulatory responses to mitigate development pressures on the natural environment. However, the major finding of this study is that this system is not likely to prevent further biodiversity loss. At least in the short term, the effectiveness of institutional responses will decrease, while the current incentive systems will accelerate land clearing and, consequently, the rate of biodiversity loss. The future trajectory will depend on the fluctuations in the political environment and the type of values society assigns to biodiversity.
7.4 Limitations and future research directions

The analysis of institutional designs or design research is an evolving field of studies. This study attempted to respond to major gaps encountered in the literature and proposed several theoretical, conceptual and analytical solutions. However, it was also recognised that exploring more fully many of the problems encountered during the research process was far beyond the scope of a single study. Therefore this chapter concludes with a brief discussion of the major limitations and suggestions for further research directions identified in both parts of the study.

**Institutional analysis**

The major limitations in institutional analysis arise from two main problems which, as Alexander (2005) points out, are common to design research. First, studies of institutional designs, especially those dealing with complex environmental problems, are comparatively new and therefore lack a sound theoretical and conceptual basis. It was found that many studies struggle with distinguishing between institutional and non-institutional factors, and with understanding what implications particular problem attributes may have for institutional designs. Second, the very nature of institutional designs creates significant challenges for the development of a sound theoretical basis. Currently practice is ahead of theory. There is a distinct gap between the general insights into institutional dimensions published in the academic literature and the work of practitioners developing complex regulatory systems. While these problems cannot be fully resolved, improvement in several areas could be made to enable progress in the evaluation of institutional performance and institutional environmental performance.

On the conceptual level, confusion and inconsistency in the use and understanding of the term ‘institutions’, in both academic and political literature, are well known problems emphasised by many analysts (Sharpf 1997, Goodin 1998a, Ostrom 2005, Dovers & Hezri 2010). Persistence of this problem in the social and environmental sciences literature was reconfirmed by this study. As the review revealed, inconsistency and conceptual confusion can be encountered even in such well known environmental policy documents as the Biodiversity Convention guidelines (SCBD 2004) and *Millennium Ecosystem Assessment* (MEA 2005b). While a single definition of the term probably will never be provided, a common understanding of ‘institutions’ as systems of rules, and a distinction between institutions and organisations, would largely advance analysis of the institutional dimension of environmental problems.

In the 1990s the environmental sciences experienced a ‘management turn’ which led to the introduction of new concepts and approaches in the resource management field. While ecosystem management has been discussed in the institutional context (e.g. Cortner et al. 1998), conceptual confusion and limited progress in the literature suggests that the ‘institutional turn’ has not yet occurred. Problems with the institutional dimension can perhaps be linked to the
need to integrate distinct schools of thought, and application of terms, in the social and environmental sciences. While this study cannot suggest a particular pathway for integration, one of the most promising concepts seems to be ‘institutional fit’ (Young 2002, Folke et al. 2007). This concept could provide an umbrella for a number of analytical themes and theories (see Cox 2012 for a discussion).

On the level of theory, the main problem lies, not in the lack of understanding of ‘what institutions mean’, but rather in a lack of integration of the large body of knowledge generated about institutions and their causal role. Reconciling different disciplinary perspectives on institutional theory or the theory of institutional design (Goodin 1998b) is a complex task. This study proposed a new typology of institutional functions. However, it is not in a position to develop a new theory of institutional causality. It could be argued that, without a common theoretical platform, confusion regarding institutional and non-institutional factors of examined problems will continue to persist.

From the methodological perspective, the main problem that still needs to be resolved is how to approach conceptualisation of institutional performance and its effectiveness. The question that remains to be answered is whether institutions, as ‘mental constructs’ or ‘design products’, should be considered as an assessment object in their own right, or if they ‘matter’ only in cases when they manifest as a behavioural response. This study took a position that ‘rules of the game’ matter, notwithstanding whether the game itself has yet been played or is finished. Therefore, it was argued that institutional effectiveness can be evaluated, not only in terms of produced outcomes, outputs or impacts, but also in terms of the potential to produce behavioural change. To some extent this approach resembles the so called ‘evaluability assessment’ distinguished in the context of social program evaluation (see Rossi et al. 2004), even though it contradicts a more widely applied managerial or results-based interpretation of effectiveness (e.g. Vedung 2009).

Another methodological gap, which has also been identified previously (McGrath 2010), is in the analysis of complex institutional systems. Numerous books have been written on the evaluation of specific policy interventions targeting particular behaviours. These approaches, however, are insufficient when the analysis requires an understanding of such complex problems as degradation of the coastal zone, or reduction of the vegetation cover at the state scale. Analysis of complex institutional systems is required just as much as program and policy evaluation. However, guidelines for modelling the system structure, rule scoping, presentation of data, and analytical techniques are insufficiently developed. These problems are evident even in such policy documents as the national and Queensland’s state of environment reports, which struggle with finding a common framework for the analysis of institutional responses.

Finally, evaluation of institutional environmental performance requires mixed methods of scientific enquiry. The social sciences are well equipped with methodological approaches
supporting evaluation of social or economic dimensions of programs and policies, but these are insufficient to address the environmental dimension. Methodologies supporting new analytical themes, such as how to measure spatial fits or misfits, how to evaluate underpinning theoretical models, or how to determine which functional linkages are addressed in regulations, are only in the early stage of evolution (e.g. Ekström & Young 2009). To fully interpret the rules, these methodologies will need to be combined with those applied by other disciplines of the social sciences, such as law or economics. How to address this challenging task, and how to link distinct theoretical frameworks, models and methods, also remains to be explored.

**Queensland Planning System**

The study examined the problem of biodiversity loss and its institutional drivers in the state of Queensland. It tried to consolidate and extend understanding of the nature and role of different institutional mechanisms providing for ‘on ground’ environmental outcomes, including those driving the behaviour of major governance actors responsible for environmental regulation. It was recognised that the overall system is highly complex and many aspects were traced only superficially. The study was also hindered by several gaps in current studies which are briefly outlined below.

The study argues that far more important than debating what is ‘effective’, or how to define ‘a rule’, is the question of how to understand and describe the system. Biodiversity loss is a widely acknowledged problem. It is relatively easy to argue that the overall system is ineffective, and to provide a generic recommendation that ‘institutional change’ is required. Yet, to provide adequate recommendations regarding *what should be changed*, evaluators need to understand *what exactly* is not working and *why*. Furthermore, they need to understand *why* a particular ‘game’ was created in the first place, and what social needs it is addressing. Every governance system has unique attributes and, therefore, recommendations derived from academic literature may not provide the best solution.

Institutional systems operating at such scales as the state are highly complex. Such outcomes as biodiversity loss are an ‘aggregated’ result of many interacting sub-systems of rules, including non-environmental (e.g. export markets, GSP measures, taxation system, fiscal strategies, voting). In practice, mapping an overall ‘ecosystem’ of rules is as complex a task as modelling trophic interactions in the natural environment. To this end, one of the biggest problems encountered in this study was how to unpack and describe the institutional setting. Many valuable studies have been done on the operation of separate governance actors (e.g. natural resource management bodies, local governments), and particular groups of instruments such as environmental laws, property rights and resource markets. At the same time, they tend to focus on the description of selected parts, without examining their role (i.e. ‘weight’) within an overall system and linkages to other components. This made the task of finding the main institutional drivers, and tracing their causal relations, very difficult.
Problems with understanding causality can be further linked to a lack of some type of environmental governance guidebook\(^45\) (cf. parliamentary handbooks). Many detailed government reports and guidelines have been produced on particular aspects of the problem, but this information is scattered and its compilation is very time-consuming. A comprehensive description of the governance system, and its (fundamental) institutional drivers, would be of assistance to analysts examining particular policies and programs.

### 7.5 Concluding remarks

This study started with an interest in an environmental problem which, on the surface, seemed to be clear. Instead, continuous refinement of the study led to the recognition and appreciation of the complexity of social systems and *how much* institutions matter in this setting. This intangible phenomenon has allowed humans to create organised systems operating at the scale of the planet. Mistakes in institutional design, however, are costly. ‘Prosperity games’ have led to the degradation of the natural environment to the extent of threatening the ability of the planet to sustain the human population. Understanding the rules, and critically reflecting on the games we all are playing, is an important step towards the resolution of the problems we have allowed to develop.

\(^{45}\) Note: the book incorporating analysis of different aspects of Australia’s environmental governance was published in 2003 by Dovers and Wild River (eds).
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## Appendix 1 - Definitions

<table>
<thead>
<tr>
<th>Term</th>
<th>Description</th>
<th>Source</th>
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<tbody>
<tr>
<td><strong>Biodiversity</strong></td>
<td>the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems</td>
<td>Biodiversity Convention, Article 2</td>
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| **Biodiversity**     | conservation: consumptive and nonconsumptive use without complete destruction/conversion  
loss: the long-term or permanent qualitative or quantitative reduction in components of biodiversity and their potential to provide goods and services  
preservation: protection of biodiversity from human interferences  
protection: biodiversity conservation and preservation | Redford and Richter (1999:1247)  
Biodiversity Convention (COP 2004, Decision VII/30)  
adapted from Redford and Richter (1999: 1247) Thesis, p.48 |
| **Ecosystem**        | dynamic complex of plant, animal and microorganism communities and the nonliving environment interacting as a functional unit                                                                                         | Biodiversity Convention, Article 2          |
| **Ecosystem**        | approach: a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way  
management: a management process which integrates scientific knowledge of ecological relationships within a complex sociopolitical and values framework toward the general goal of protecting native ecosystem integrity over the long term management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and processes necessary to sustain ecosystem structure and function | Biodiversity Convention (COP 2000, Decision V/6)  
Grumbine (1994:31)  
Christensen et al. (1996:668-9) |
| **Ecosystem**        | services: benefits that ecosystems provide                                                                                                                                                            | MEA (2005a:27)                              |
| **Effectiveness**    | institutional: the extent to which an institution influences or can potentially influence the behavioural response of targeted actors towards the outcomes an institution has been designed to produce  
institutional environmental performance: the extent to which an institution influences, or can potentially influence, the behavioural response of targeted actors towards the achievement of the environmental effects an institution has been designed to produce | Thesis, p.30  
Thesis, p.39 |
<p>| <strong>Environmental</strong>    | governance: a set of regulatory processes, mechanisms and organisations through which political actors influence environmental actions and outcomes                                                                 | Lemos and Agrawal (2006:298)                |</p>
<table>
<thead>
<tr>
<th>Institutions</th>
<th>systems of established and prevalent social rules that structure social interactions</th>
<th>Hodgson (2006:2)</th>
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<tr>
<td></td>
<td>the prescriptions that humans use to organise all forms of repetitive and structured interactions</td>
<td>Ostrom (2005:3)</td>
</tr>
<tr>
<td></td>
<td>the rules of the game in a society or, more formally […] the humanly devised constraints that shape human interactions</td>
<td>North (1990:3)</td>
</tr>
<tr>
<td></td>
<td>formal systems of rules that are articulated in constitutive documents such as laws, regulations, agreements or other policy instruments</td>
<td>adapted from Young (2002:5-6)</td>
</tr>
<tr>
<td></td>
<td>informal systems of rules that structure human interactions and are embedded in established practices and informal understanding regarding appropriate behaviour</td>
<td>adapted from Young (2002:6)</td>
</tr>
<tr>
<td>Institutional</td>
<td>an institutional influence on, or contribution to, the behavioural response of targeted actors, which either directly or indirectly produces, or has a potential to produce, environmental effects (impacts)</td>
<td>Thesis, p.39</td>
</tr>
<tr>
<td>environmental</td>
<td>goals statements specifying the (set of) effects an institution has been designed to produce</td>
<td>Thesis, p.38</td>
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<tr>
<td>performance</td>
<td>impacts changes in the state of the biophysical environment resulting from the implementation of rules</td>
<td>adapted from Underdahl (2002:6)</td>
</tr>
<tr>
<td>outcomes</td>
<td>changes in human behaviour resulting from the introduction of rules</td>
<td>adapted from Underdahl (2002:6)</td>
</tr>
<tr>
<td>outputs</td>
<td>plans, projects, and other tangible items, including institutions produced as part of the implementation of rules</td>
<td>adapted from Underdahl (2002:6)</td>
</tr>
<tr>
<td>performance</td>
<td>an institutional influence on, or contribution to, the behavioural response of targeted actors</td>
<td>Thesis, p.29</td>
</tr>
<tr>
<td>performance</td>
<td>performance criteria normative statements specifying what aspects (effects) of institutional performance are evaluated</td>
<td>Thesis, p.33</td>
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<tr>
<td>dimension</td>
<td>performance dimension configurations of rules contributing to the behavioural response of targeted actors which produces particular effects</td>
<td>Thesis, p.33</td>
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<tr>
<td>performance</td>
<td>performance reference point counterfactual point to which observed outcomes can be compared to identify institutional influence</td>
<td>Mitchell (2008:81)</td>
</tr>
<tr>
<td>reference point</td>
<td>performance standard normative points to which observed results can be compared to assess the magnitude of institutional influence or success or failure or satisfactory performance</td>
<td>adapted from Mitchell (2008) and Vedung (2006)</td>
</tr>
<tr>
<td>Landscape</td>
<td>an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors</td>
<td>European Landscape Convention, Article 1</td>
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<tr>
<td>Organisation</td>
<td>groups of individuals bound by some common purpose to achieve objective</td>
<td>North (1990:5)</td>
</tr>
<tr>
<td>Organisational rules</td>
<td>systems of rules established within an organisation to structure interactions of its members</td>
<td>Thesis, p.21</td>
</tr>
<tr>
<td>Species</td>
<td>a group of actually and potentially interbreeding individuals</td>
<td>Krohne (2001:7)</td>
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