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Advancing land-sea conservation planning: cross-system threats and competing values

PhD thesis submitted by
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May 2012

For the degree of Doctor of Philosophy
Australian Research Council Centre of Excellence for Coral Reef Studies, James Cook University
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Abstract

Overfishing, pollution, coastal development and climate change threaten marine biodiversity globally and compromise the services that marine ecosystems provide. Spatial management, including setting aside land for conservation, is central to curbing the global decline in biodiversity, but many threats originate from beyond the boundaries of protected areas. This is a particular problem in marine systems, which are influenced by many activities on land. Integrating land and sea conservation planning is therefore of utmost importance. Systematic conservation planning (SCP) provides a framework to identify areas where actions can be effective in addressing these threats, while minimising the costs of interventions. Methodologies to incorporate land-sea connections in conservation planning are developing quickly, but most conservation planning initiatives have not embraced an integrated approach. Recent advances in theories and tools in conservation planning can contribute to overcoming these limitations and explicitly incorporating land-sea connections. However, additional scientific knowledge is needed to address cross-system threats and to manage the complexities of conservation across realms.

The goal of my thesis is to improve understanding of fundamental elements in land-sea conservation planning and investigate how these can be incorporated into the planning process. More specifically, my project explores theoretical and methodological aspects of two decision-making problems associated with a land-sea approach: integrating cross system-threats (especially considering how land-based threats can affect marine and coastal spatial prioritisation), and considering competing values (i.e., spatial incongruence between terrestrial (upstream) and marine (downstream) values of coastal catchments). Three objectives bound the scope of my thesis: (1) to assess progress in land-sea planning theory and practice, and identify key elements to achieve land-sea integration, and the challenges thereof (Chapters 2 and 3); (2) to develop novel methods for incorporating cross-system threats, especially mapping and targeting land-based threats (Chapters 4 and 5); and (3) to explore coincidence between land values derived from terrestrial and marine conservation objectives (Chapter 6).

In order to address these objectives, first I reviewed the literature describing connections between land and sea, and how they have been incorporated into conservation planning (Chapter 2). Key results of this chapter include: a classification of land-sea processes, which also identifies potential planning strategies (design criteria, variable objectives, planning units, and movable areas) to incorporate these processes into planning. I also identified major cross-system threats and elements that planners should consider (sources, affected realms, directionality of influence, stressors, and areas for intervention). The main output of this chapter is an operational framework to guide land-sea planning informed by expert opinion and case studies that considered land-sea connections.

I then analysed marine planning initiatives in the Gulf of California (GoC) to investigate whether the elements identified in the framework have informed practical applications (Chapter 3). The GoC is an ideal case study for comparing areas identified as priorities in multiple marine conservation plans, with seven marine planning exercises undertaken in the past 15 years. Despite some convergence, I found important spatial differences in priorities between plans. The existence of multiple marine conservation plans in the GoC highlighted some of the complexities and benefits of having multiple sets of priorities. I also showed that the use of SCP methods has progressed slowly and I highlighted benefits and difficulties of applying SCP principles and tools.

A major gap in marine management of the GoC is the need to address land-sea connections, particularly land-based pollution which has been identified as a key threat to marine ecosystems. I used catchment modelling to assess the magnitude of the problem and to identify areas that require intervention for mitigation of land-based threats (Chapter 4). Different catchment modelling approaches to identify sources of nutrients and sediments vary in data requirements, outputs, and applications. I compared two catchment models commonly used to target catchment management (N-SPECT and SedNet) in terms of their approach (method and assumptions), data requirements (availability and preparation), performance (differences between modelled and observed runoff and suspended sediment loads), and outputs (similarity in spatial pattern of pollutant supply and end-of-river loads). I found that both models are generally applicable under limited data circumstances, but SedNet requires additional datasets and parameters. The patterns of supply are similar and outputs of both models are adequate

for prioritising long-term management in land-sea planning applications. However, the lower performance of N-SPECT limits its application for plume and ecological modelling. Additional functionality in SedNet offers useful outputs for targeting catchment management. This includes accounting for the effects of reservoir trapping of suspended sediment.

In addition to mapping sources of land-based pollution, identifying the affected areas and the scale of influence on marine ecosystems is critical to assess the ecological impacts of degraded water quality. I developed an improved model of exposure to river plumes (based on satellite imagery) to assess the potential impact of land-based pollution, using the Great Barrier Reef (Australia) as a case study (Chapter 5). To build the model, first I mapped plumes and identified surface water (colour) classes using maximum-likelihood classification of enhanced satellite images. Second, I summarised weekly data and calculated annual frequency of plume occurrence and mean colour class values. Third, I calculated the proportional contribution of major rivers to the regional total suspended sediment (TSS) and dissolved inorganic nitrogen (DIN) loads. Following that step, I scaled the proportional contributions with a cost-distance function (known plume extents, peak flows and mean colour class). Finally, I calculated exposure by combining scaled contributions and frequency of occurrence. I then used the exposure model to quantify the extent of each exposure category (TSS/DIN) and the affected area of coral reefs and seagrass beds. I found that classification of true colour satellite images can be used to map plumes and to qualitatively assess exposure to river-borne pollutants. The model I developed is useful to study the spatial and temporal variation in exposure of coastal-marine ecosystems to riverine plumes. The observed inter-annual variation in exposure of habitats to pollutants stresses the need to incorporate the temporal component in exposure and risk models.

In Chapter 6, I used the outputs from the catchment model and a river plume models to develop an integrated land-sea plan for selected catchments in the GoC. I used the catchment model to map sources and loads of selected pollutants (TSS and TN). I also used a land use change model to calculate the probability of change to anthropic land use classes and the vulnerability of natural vegetation to clearing. Based on these two outputs, and the link between pollutant sources (catchments) and the affected marine areas (of varying conservation priority), I set two types of objectives to prioritise

management of land areas: biodiversity conservation (protection of vertebrates and vegetation types, and aimed to target areas with a higher vulnerability to be converted into anthropic areas) and water quality objectives for catchment management (based on contribution to pollutant loads, vulnerability to clearing, magnitude of change from natural conditions and link to marine priority conservation areas). My results emphasise that integrating land use change in spatial prioritisation procedures is needed to better inform management for terrestrial conservation and water quality. Integrating catchment and river plume models is thus critical to assessing effectiveness of management actions to achieve water quality objectives.

In summary, my thesis advances conservation planning in five ways: (1) through the proposal of an operational framework that identifies and integrates elements crucial to integrated land-sea conservation planning; (2) by providing insights into convergence of priorities derived from different planning approaches and benefits and difficulties of having multiple sets of plans; (3) by identifying key factors influencing the selection of catchment models for regional land-sea planning applications; (4) by developing an improved plume exposure model to inform catchment management and design of MPA networks; and (5) by providing a method for integrating catchment, land use change and plume models for land-sea planning applications.

Outputs associated with this thesis

Peer-reviewed papers[1]

Ćlvarez-Romero, J. G., R. L. Pressey, N. C. Ban, K. Vance-Borland, C. Willer, C. J. Klein, and S. D. Gaines. 2011. Integrated land-sea conservation planning: the missing links. *Annual Review of Ecology, Evolution, and Systematics* **42**: 381-409

Ćlvarez-Romero, J. G., R. L. Pressey, N. C. Ban, J. Torre-Cosio, and O. Aburto-Oropeza. In review. Marine conservation planning in practice: lessons learned from the Gulf of California. *Aquatic Conservation: Marine and Freshwater Ecosystems*

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[1] This thesis is based on the following manuscripts published, in review or in preparation which have been edited here to reduce redundancy and ensure a consistent terminology.

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Conference presentations and posters

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Jul 11-16, 2009: *23rd Annual Meeting of the Society for Conservation Biology / 2009 International Conservation Biology Congress*. Oral presentation: **J. G. Ćlvarez-Romero** and R.L. Pressey, Planning from the land to the sea: an integrated operational framework, In: Symposium ñIntegrated protection and restoration of wetlands, rivers and subterranean aquatic ecosystemsò. Beijing, China.

May 11-15, 2009: *World Ocean Conference 2009 and Coral Reef Triangle Initiative Meeting*. Poster: D. Freitas, R.L. Pressey, **J. G. Ćlvarez-Romero**, S. Januchowski and J. VanDerWal, Resolving trade-offs between upstream and downstream conservation objectives in coastal catchments. Manado, Indonesia.

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Boyle, M. J., **J. G. Ćlvarez-Romero**, and T. P. Hughes. 2008. *Ningaloo Coast World Heritage Nomination: A comparative analysis with similar properties, Final Report*. Australian Research Council Centre of Excellence for Coral Reef Studies, James Cook University, Townsville, Australia, 65 + 22 pp.

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Glossary

Conservation areas: Places where some form of management, ranging from strict reservation to off-reserve management, best-practice uses, and restoration, is undertaken to contribute to conservation objectives.

Conservation context: Governance, current ownership, uses, existing management, and threats that make up the context for a planning exercise, including interactions between terrestrial and marine socio-ecological systems.

Conservation objectives: Specific interpretations of goals through the filter of available data; preferably quantitative (otherwise qualitative) statements about how much of each feature (e.g. habitat types, species, ecological processes) should be represented in conservation areas. Objectives can be expressed in terms of percentage of area, number of occurrences or abundances of individuals, or relate to configuration, past disturbance and other criteria.

Costs: Relevant socioeconomic costs include acquisition costs, management costs, transaction costs, and opportunity costs. Costs are often expressed, not in dollar terms, but as the suitability of areas for conservation management, integrated across criteria that reflect constraints and opportunities.

Cross-system threats: Threats to biodiversity that originate in one realm and affect another.

Decision support tools: Conservation planning software (e.g., Marxan, C-Plan) and other tools (e.g., for catchment modelling and land-use planning) that provide analytical and graphical capabilities to facilitate negotiation, exploration of scenarios, and the involvement of experts and other stakeholders.

Ecological processes: Sequences of changes in biological and physical characteristics, including migrations, metapopulation dynamics, dispersal, and flows of non-living materials; ecological processes operate at many spatial and temporal scales.

Ecosystem services: Resources and processes supplied by natural ecosystems for the benefit of humans. Ecosystem services include: provisioning, such as the production of food and water; regulating, such as the control of climate and disease; supporting, such as nutrient cycles and crop pollination; and cultural, such as spiritual and recreational benefits.

Goals: A collective vision of aspirations, including, for example, the representation or persistence of biodiversity, protection of endangered species, enhanced natural resources management, improved livelihoods for communities, maintenance of ecosystem services, preservation of cultural heritage, and safeguarding of traditional knowledge.

Irreplaceability: A measure of the importance of areas in contributing to conservation objectives. Units with higher irreplaceability have fewer possible replacements and are more likely to be needed for inclusion in a system of conservation areas to achieve objectives. Maps of irreplaceability are also maps of options for achieving objectives, giving scope for participation and negotiation.

Mainstreaming: The translation of technical planning products into various forms readily accessible to a diversity of stakeholders and sectors of society to facilitate the implementation of conservation actions.

Marine protected area (MPA): According to the IUCN, an MPA is any area of intertidal or sub-tidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment. MPAs include different types of protection, including no-take zones (e.g., to enable fish stocks to recover) and multiple use of their resources.

Planning domain: Region across which conservation areas are being assessed and compared for investment in actions to achieve conservation objectives.

Planning units: Spatial subdivisions of planning domains that are the units of assessment and comparison in conservation planning. Planning units can be natural, administrative, or arbitrary subdivisions of land/seascapes that form the building blocks for systems or networks of conservation areas.

Stakeholders: Communities, natural resource users, governmental agencies, scientists, local experts, and NGOs who will implement or be affected by conservation actions or who could contribute information and expertise to the planning process.

Systematic conservation planning: A process to guide the spatial allocation of limited resources to achieve explicit, and mostly quantitative, conservation objectives. The defining elements of systematic conservation planning include complementarity between selected areas, least-cost solutions to achieving objectives, and transparent and repeatable methods for designing configurations of conservation areas.

Thesis chapters

Chapter 1. General introduction

Rationale

This thesis aims to advance fundamental elements of land-sea conservation planning. Below I highlight and briefly synthesise literature from several topics that culminated in the focus of my thesis. These are: threats to marine biodiversity, actions to mitigate biodiversity loss, guiding efficient use of conservation resources, threats beyond boundaries of conservation areas, and ecological connections between land and sea. Together, these topics illustrate the need for integrated land-sea planning. Only a short literature review is provided here because Chapter 2 provides an in-depth review.

Threats to marine biodiversity

Marine ecosystems provide many services to people, including provision of food, mitigation of coastal hazards, and recreation (CBD 2006; Granek et al. 2010), but the delivery of these services is threatened by accelerated habitat degradation and unprecedented population declines of many marine species (Gray 1997; MEA 2005; Worm et al. 2006). Human impacts on the worldôs oceans can be traced globally to fishing, pollution and climate change (Halpern et al. 2008; Worm et al. 2009). Threats are generally higher in coastal and near-shore areas, where coral reefs, mangroves and rocky intertidal communities are amongst the most vulnerable ecosystems (Halpern et al. 2007). In most cases, marine systems are threatened by a multiplicity of stressors (e.g., fishing and UV radiation) and experience cumulative impacts (Mullan-Crain et al. 2008). This fast growing and widespread degradation of marine resources and ecosystems demands effective mechanisms to conserve the biodiversity and services that these systems provide (Barbier et al. 2010; Granek et al. 2010; Lubchenco et al. 2003).

Actions to mitigate biodiversity loss

Several kinds of actions have been implemented to address the multiple threats affecting coastal and marine ecosystems, including improving use of marine resources (e.g., improved regulation of fisheries), minimising impact of marine and coastal activities (.e.g., regulating transport and coastal development), and managing specific areas of high conservation value. A common conservation strategy is the implementation of marine protected areas (MPAs). MPAs are an essential component of conservation strategies that aim to promote marine biodiversity from the local to the global scales (Carr et al. 2003; Fox et al. 2012). MPAs have proven to be an effective tool for marine conservation and management, with documented benefits both within and outside their boundaries (Lester et al. 2009; Lubchenco et al. 2003; Roberts et al. 2001a; Wood et al. 2008). A key consideration when placing MPAs is their role in minimising threats to the conservation features within them or avoiding areas of high use/threat (Wilson et al. 2005a).

Guiding efficient use of conservation resources

A critical question derived from the need to increase our conservation efforts to halt biodiversity loss is: Where should we place conservation areas to effectively achieve conservation objectives? Limited conservation resources and social and political constraints restrict the number and extent of areas that can be managed (Sarkar et al. 2006). Systematic conservation planning (SCP) provides a framework to identify areas where conservation actions effectively achieve conservation objectives, while minimising the costs of interventions (Margules & Pressey 2000). SCP provides a framework to guide decisions in a transparent and repeatable fashion, and provides decision-makers with maps of options that can be assessed with stakeholders in terms of their contribution to objectives and the cost of implementation (Pressey & Bottrill 2009). The value of this approach is reflected on the increasing application of SCP methods and tools to guide the design and zoning of MPAs worldwide, some of which have been successfully implemented or in the process to be implemented (e.g., Great Barrier Reef Marine Park, Australia: Fernandes et al. 2005; California MPA network: Gleason et al. 2010; Prince Edward Island, South Africa: Lombard et al. 2007). Marine

planning has advanced rapidly (Leslie 2005), including planning for pelagic processes (Game et al. 2009), incorporating risk to natural catastrophes (Allison et al. 2003), considering the potential effects of climate change (Green et al. 2009), and maintaining biological connectivity between MPAs (Almany et al. 2009).

Threats beyond boundaries of conservation areas

Conservation areas are vulnerable to natural resource development and exploitation occurring outside their boundaries. This is particularly relevant for MPAs because of the more open nature of marine systems (Avery 2003; Carr et al. 2003) and the asymmetrical connection ï mostly downstream ï between the land and sea (Stoms et al. 2005). Activities occurring within coastal catchments, sometimes hundreds or thousands of kilometres away from the coasts (e.g., Gulf of California: Beman et al. 2005; Great Barrief Reef: Brodie et al. 2010; Gulf of Mexico: Mitsch et al. 2001), are altering flows of water, sediments and nutrients, or adding pollutants that reach nearshore and offshore areas (Halpern et al. 2008). Worldwide, agriculture, grazing, logging and urbanisation continue to expand, hence the impacts of land-based pollutants associated with these activities have intensified. Coastal ecosystems are particularly vulnerable to these threats due to elevated population densities living along the coasts and high inputs of contaminants derived from human activities in coastal zones (Letson et al. 1998).

While MPAs are generally planned to separate elements of biodiversity from processes that threaten their persistence (Margules & Pressey 2000), impacts of pollution, natural catastrophes, climate change, and over-fishing cannot be alleviated by MPAs alone and should be complemented by other approaches (Leslie 2005; Lubchenco et al. 2003; Roberts et al. 2003a). Most MPAs are located in coastal waters (WDPA 2010). In many cases external threats to MPAs have greater impacts than the activities within their boundaries (Cicin-Sain & Belfiore 2005). Therefore, most of the biodiversity and fisheries benefits that MPAs provide are undermined by poor management and the lack of attention to external threats (Wood et al. 2008). These have been referred to as cross-system threats. The spatial configuration of marine conservation priorities can change if these threats are considered (Tallis et al. 2008).

Ecological connections between land and sea

Flows of materials and organisms define biological and physical processes that operate across the land-sea boundary, and sustain biodiversity across the terrestrial, freshwater and marine realms (Beger et al. 2010b; Polis & Hurd 1996; Wipfli et al. 2007). These processes depend on uninterrupted connections and natural flows, as well as on the integrity of their components in the different realms (Pringle 2006). Examples include fertilisation of terrestrial and nearshore coastal marine areas with marine-derived nutrients mediated by pelagic predators (e.g., seabirds, pinnipeds) feeding in offshore areas (Anderson & Polis 1999; Lombard et al. 2007), input of marine-derived nutrients into terrestrial and freshwater ecosystems (Helfield & Naiman 2006; Naiman et al. 2002), delivery of nutrients and organic material from forests to estuaries (Naiman & Sibert 1978; Wipfli et al. 2007), and movements of animals utilising both terrestrial and marine habitats (Burger et al. 2000; Vasquez-Montoya & Thorson 1982).

Ecological interactions between the terrestrial, freshwater, and marine environments have been recognised and studied for a long time, especially those between adjacent terrestrial and marine coastal ecosystems, but an emphasis on these linkages for marine conservation is only recent (Abell et al. 2007; Letson et al. 1998; Sloan et al. 2007). Furthermore, these land-sea interactions are usually not explicitly considered in marine conservation planning exercises (Stoms et al. 2005). Thus, in addition to cross-system threats, conservation area design should also consider the persistence of ecological processes that depend on continuity of flows between the marine and terrestrial realms (Beger et al. 2010b). Maintaining the flow of nutrients between land and sea, as well as conserving land and freshwater areas utilised by marine species (and *vice versa*) can also influence the location of conservation areas (Hazlitt et al. 2010).

Research gaps

While links between land and sea realms are widely recognised, the methods to accomplish integrated land-sea planning are not yet fully developed and hence such integrated planning remains elusive. During the past few decades, research and practice in conservation biology has produced the fundamental principles, methods, and tools of systematic conservation planning. Based on these, conservation planners have designed

networks of protected areas and implemented conservation actions by identifying the areas that are representative of the biodiversity of a region and that can promote the long-term persistence of this biodiversity and the processes that sustain it (Groves et al. 2002; Margules & Pressey 2000; Pressey et al. 2007). Following these principles, more than 90 planning exercises have identified conservation areas in the marine environment, but only a few of these considered aspects that are relevant to land-sea integration, such as protecting habitats used by land and marine species or maintaining clean water provision (Leslie 2005). Very few studies have considered external threats or processes that occur across the land-sea interface and most identify conservation areas in the terrestrial and marine realms independently (Beck 2003; Roberts et al. 2003a). Thus, given marine biodiversity declines and the importance of land-sea interactions and cross-system threats, a major research gap is integration of land and sea in conservation planning. This thesis is an attempt to further that endeavour.

Thesis goal and objectives

The overarching goal of my thesis is to improve understanding of fundamental elements in land-sea conservation planning, and investigate how these can be incorporated into the planning process. I address this goal through three specific research objectives:

1. Assess progress in land-sea planning theory and practice, and identify landmarks and challenges;
2. Develop novel methods for incorporating cross-system threats: mapping and targeting land-based threats; and
3. Explore coincidence between land values derived from terrestrial (within catchment) and marine (downstream) conservation objectives.

Thesis structure

In addition to this introduction (Chapter 1) and the general discussion (Chapter 7), I furthered my research objectives through five chapters, which are presented as a series of research chapters formatted for publication in peer-reviewed journals. **Figure 1.1** shows the overall structure of the thesis. Tables and figures are shown throughout the

text and additional supporting materials for some of the chapters are provided in the appendices.

Figure 1.1. Chapter structure for this thesis. Numbers in the orange boxes correspond to research chapters, and the numbers on the right of the green boxes to the research objectives that each chapter addresses.

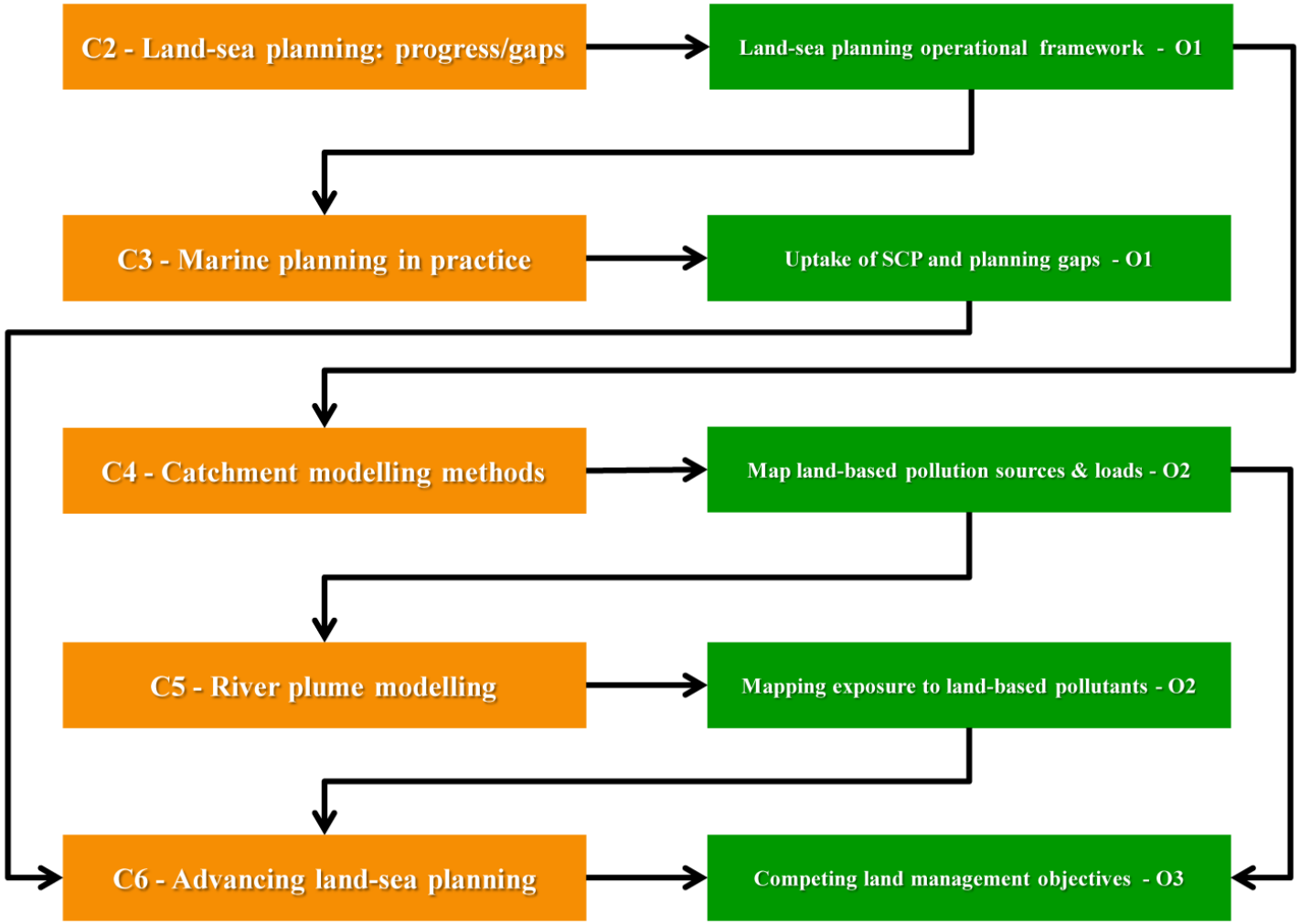

Figure 1.1 Chapter structure for this thesis

Chapter 1 (this chapter) provides the rationale for the focus of the thesis through a brief summary of the theoretical and practical context, and outlines the structure of the thesis.

Chapter 2 addresses Objective 1 by reviewing existing land-sea planning exercises. The research questions addressed in this chapter are: What are the ecological and socioeconomic couplings between land and sea relevant to planning? How can conservation science inform more effective integrated land-sea planning? And what is our current capability in integrated land-sea planning and where are the major research gaps? To answer these questions, I review literature describing connections between land and sea, and how they have been incorporated into conservation planning. I also analyse 26 case studies (planning exercises), which have taken into consideration land-sea integration. Based on the approaches and findings of these studies, I develop an operational framework aimed to guide integrated land-sea conservation planning. Finally, using the framework as a reference, I interviewed experts to identify which aspects have been addressed and which are the most critical gaps.

Chapter 3 addresses Objective 1 through a comprehensive analysis of marine planning in the Gulf of California, including an assessment of progress in the use of SCP principles/tools, the spatial and methodological similarity between plans, and the uptake of planning outputs in terms of implementation of spatial actions. The research questions addressed in this chapter are: what are the gaps and difficulties of applying systematic conservation planning methods? How similar are the data and methods employed, and how similar are the subsequent spatial plans? What are the benefits and difficulties of having multiple conservation plans for the same area? To answer these questions I use the operational framework developed in chapter 2 to assess methodological similarities and differences and identify planning gaps. I also assess similarity and coincidence between plans in terms of their methods (focused on objectives and data, because I consider these the most influential in selection of priority areas), spatial similarity of priority areas (as well as a comparison of spatial configuration of priority maps), coincidence of priority areas with key features (overlaps of priority areas with existing MPAs, marine ecoregions, and selected ecosystems and seascape features). I also interviewed planners to assess achievement of goals, how planning outputs have influenced spatial actions, and factors facilitating or constraining implementation.

Chapter 4 addresses Objective 2 through a comparison of two catchment models, in particular to assess their potential application for water quality management in coastal-marine planning exercises. The research questions addressed in this chapter are: How do catchment models differ in regards to data requirements, ease of implementation, accuracy and outputs? How do these factors influence the selection of models to inform regional-scale decisions for catchment management? To answer these questions I use the Gulf of California as case study, where the land-based threats are documented and the need to integrate cross-system threats is recognised as a priority. I compare two catchment models commonly used to target catchment management (N-SPECT and SedNet) in terms of their approach (method and assumptions), data requirements (availability and preparation), performance (differences between modelled and observed runoff and suspended sediment loads), and outputs (similarity in spatial pattern of pollutantsôsupply per sub-catchment and end-of-river loads). Based on the previous elements, I discuss the pros and cons of models for land-sea planning applications.

Chapter 5 addresses Objective 2 through a novel approach to model exposure of coastal marine ecosystems to land-based pollutants based on satellite imagery. The research questions addressed in this chapter are: Can true-colour classification of satellite images be used to map plumes and better understand pollutant dispersal? How do temporal changes in river discharge and pollutant loads affect exposure of coastal-marine habitats to riverine plumes? To answer these questions I use the Great Barrier Reef Marine Park (GBR), Australia, as case study to develop the exposure model. To build the model, first I map plumes and identify surface water (colour) classes using maximum-likelihood classification of enhanced satellite images. Second, I summarise weekly data and calculate annual frequency of plume occurrence and mean colour class values. Third, I calculate the proportional contribution of major rivers to the regional total suspended sediment (TSS) and dissolved inorganic nitrogen (DIN) loads. I then scale the proportional contributions with a cost-distance function (known plume extents, peak flows and mean colour class). Finally, I calculate exposure by combining scaled contributions and frequency of occurrence. I then use the exposure model to quantify extent of each exposure category (TSS/DIN) and the affected area of coral reefs and seagrass beds.

Chapter 6 addresses Objective 3 by integrating the results from the catchment model (Chapter 4) and the river plume model (Chapter 5) to develop a method to prioritise catchment management for coastal-marine water quality. The research questions addressed in this chapter are: How do sources of pollutants within catchments relate to specific coastal-marine ecosystems? What are the different ways in which land-based threats to coastal-marine ecosystems can be minimised? What are the similarities and differences in the distribution of land values within catchments for terrestrial and marine conservation? To answer these questions I use several catchments in the Gulf of California as case study. I use the SedNet catchment model to map sources and loads of selected pollutants (TSS and TN). I also use a land use change model to calculate the probability of change to anthropic land use classes and the vulnerability of natural vegetation to clearing. Based on these two outputs, and the link between pollutant sources (catchments) and the affected marine areas (of varying conservation priority), I set two types of objectives to prioritise management of land areas: biodiversity conservation (protection of vertebrates and vegetation types, aiming to target areas with a higher vulnerability to be converted into anthropic areas) and water quality objectives for catchment management (based on contribution to pollutant loads, vulnerability, magnitude of change from natural conditions and link to marine priority conservation areas).

Chapter 7 provides a summary of the previous chapters and a discussion of the contributions made to advance the cross-disciplinary field of integrated land-sea conservation planning. I also highlight the opportunities and constraints of addressing the research gaps identified in Chapters 1 and 2, in particular regarding the incorporation of cross-system threats (Chapters 4-6) and competing objectives (Chapter 6) in the planning process.

Chapter 2. Integrated land-sea conservation planning: the missing links

Abstract

Spatial management, including setting aside conservation areas, is central to curbing the global decline of biodiversity, but many threats originate from beyond the boundaries of conservation areas. This is a particular problem in marine systems, which are influenced by many activities on land. In addition, connections between land and sea support many species and ecological processes valued for conservation. Integrated land and sea conservation planning is therefore of utmost importance. I review the literature describing connections between land and sea and how they have been incorporated into conservation planning. Land-sea connections include: land-sea processes, the natural flows occurring between realms; cross-system threats, which originate in one realm and affect another; and socioeconomic interactions associated with management decisions to maintain or restore land-sea processes and to prevent or mitigate cross-system threats. I highlight the need to explicitly incorporate land-sea connections in conservation planning and suggest ways of doing this through the use of a novel operational framework for integrated land-sea planning. On the basis of expert surveys and a literature review, I also identify those aspects of conservation planning for which improved integration between land and sea is most needed.

Introduction

Continued loss of biodiversity remains a major international concern on land and in the sea (Rands et al. 2010; Stokstad 2010), and many efforts to curb this decline involve spatial protection through systematic conservation planning (Margules & Pressey 2000). The history of protected areas as a formal conservation tool began on land with the establishment of Yellowstone National Park in 1872. The first marine protected area (MPA) was designated in Australia in 1879, but the pace of efforts in the sea has lagged that of those on land by several decades. Because conservation efforts were more prevalent on land, marine conservation science was built on a rich history of terrestrial theory and application (Kirkpatrick 1983; Margules & Pressey 2000). Although there are key differences in marine systems that require adaptation of terrestrial approaches

(e.g., different scales and patterns of dispersal as well as different types of threats: Halpern et al. 2008; Kinlan & Gaines 2003), the foundation of planning principles on land has played a pivotal role in the rapid advancement of marine conservation planning.

Despite shared conceptual roots, conservation planning in terrestrial and marine realms has largely proceeded as if the ecological systems were unconnected (Beck 2003; Stoms et al. 2005). This lack of integration is especially problematic in the sea because the physical and ecological connections between land and sea are often highly asymmetrical, with marine areas more influenced by land than vice versa (Stoms et al. 2005). MPAs designed to protect marine biodiversity (Wood et al. 2008) are vulnerable to both land- and sea-based threats originating outside their boundaries (Boersma & Parrish 1999; Cicin-Sain & Belfiore 2005). Because most MPAs are in coastal waters (Wood et al. 2008), the risks from diverse land-based activities are further accentuated (Halpern et al. 2009). In this review, I examine our understanding of the connections between land and ocean systems as well as how these connections should guide integrated conservation design and implementation of conservation actions.

Conservation planning focuses on the spatial allocation of conservation resources to different actions (e.g., protected areas, restoration sites, managed harvesting, and best-practice land use: Sarkar et al. 2006), which is complemented by non-spatial activities, such as legislation, policy, incentives, capacity building, and education (Pressey & Bottrill 2009). Such complementary efforts strengthen conservation areas and expand the management and sustainable use of natural resources to areas surrounding or linked to focal conservation areas.

Land-sea conservation planning incorporates ecological connections between land and sea and seeks to limit land-based threats to MPAs, but neither a comprehensive review nor an operational framework for land-sea planning exists. Much has been written about integrated coastal zone management and marine ecosystem-based management (see Cicin-Sain & Belfiore 2005; McLeod & Leslie 2009). These approaches often address aspects of land-sea planning such as ecological connections between land and sea, cumulative impacts, multiple objectives, diverse stakeholders, and jurisdictional fragmentation. However, neither addresses the fundamental concepts of systematic

conservation planning, namely, complementarity between selected areas, least-cost solutions to achieving objectives, and transparent and repeatable methods for designing configurations of conservation areas (Margules & Pressey 2000). Ideally, integrated land-sea conservation planning combines systematic principles with aspects of other approaches to delineate conservation areas and to set priorities for action across realms. Beyond the requirements for planning in a single realm (Pressey & Bottrill 2009), land-sea planning should involve explicit conservation objectives for processes that connect the land and the sea as well as explicit ways of accounting for threats that originate in one realm and affect the other. Land-sea planning also addresses the urgent need to effectively integrate conservation investments on land and in the sea to maximise benefits and minimise costs of conservation actions.

Some of the conservation implications of connections between land and sea have been examined previously. Beger *et al.* (2010a) assessed the methods, challenges, and potential for spatial analyses to incorporate processes connecting terrestrial, freshwater, and marine environments. Stoms *et al.* (2005) outlined a conceptual model and methods to incorporate land-sea linkages that either benefit or threaten marine environments. Tallis *et al.* (2008) explored potential changes to MPA design to account for threats originating on land. Klein *et al.* (2010) proposed a method to maximise the return on investment in terrestrial and marine conservation actions to minimise threats to coral reefs originating both on land and in the sea. These studies contributed novel concepts and methods to the quickly evolving and highly complex field of land-sea planning. However, none of these papers integrated all the necessary considerations for comprehensive land-sea planning, which is a primary objective of this review. In addition, non-governmental organisations (NGOs) and government agencies from around the world have adapted systematic conservation planning to address the practical difficulties of integrating land and sea. Reported mostly in the gray literature, their insights are obscure to the broader conservation community. For the first time, I review their lessons learned and contributions to theory and practice.

Land-sea connections: ecological processes, threats, and socioeconomic interactions

Land-sea ecological processes

Ecological processes are critical, although often neglected, elements in conservation planning (Cowling et al. 1999; Pressey et al. 2007). Some ecological processes link the land and the sea (hereafter referred to as land-sea processes); these are mediated by the flow of water and movements of organisms between terrestrial, freshwater, and marine ecosystems (Beger et al. 2010a) and are critical to the persistence of biodiversity across realms. Examples include river inputs of nutrients with effects on the productivity and composition of coastal marine communities (Caddy & Bakun 1994; Humborg et al. 2000; Naiman & Sibert 1978), subsidies to land food webs from marine-derived nutrients (Helfield & Naiman 2001; Naiman et al. 2002; Polis et al. 1997), and ecosystems on the margins of the two realms (e.g., mangroves) that provide protection for vulnerable life stages of terrestrial, freshwater, and marine species (Nagelkerken et al. 2008).

Using the framework of Beger *et al.* (2010a), I classified a range of land-sea processes from the scientific literature and conservation planning studies (**Table 2.1**). Although not exhaustive, this overview describes processes that are diverse, operate over a wide range of spatial and temporal scales, and include linkages in both directions between land and sea. Even though land-sea processes are relatively well documented, information about them relevant to conservation planning is generally dispersed.

Table 2.1. Examples of ecological processes linking land and sea: a) Land-sea processes can be mediated by two types of linkages: interfaces and connections. Interfaces are areas where two or more realms and their processes are intermixed. These can be either narrow (e.g., rocky intertidal communities) or broad (e.g., estuaries). Connections are linkages between two end points in different realms that are not adjacent. They can be mediated by well defined or constrained paths, such as rivers connecting coastal catchments and oceans, but can also be diffuse, without well defined movement paths of organisms or material (Beger et al. 2010a); b) the listed conservation features (i.e., elements within terrestrial, freshwater, or marine realms) are the spatial elements that can be targeted in conservation planning. Conservation actions will be most effective if these elements are identified with assistance from local experts.

		Examples of land-sea processes [a]	Conservation features [b]
CONNECTIONS	*Constrained*	Movement of euryhaline species, e.g., bull sharks, between freshwater and marine systems (Vasquez-Montoya and Thorson 1982)	Lakes, large river systems, estuaries, marine pelagic
		River input of freshwater and nutrients that influence estuarine and marine systems (Caddy and Bakun 1994, Rowell et al. 2008, Humborg et al. 2000)	Forests, major river and stream systems, estuaries, marine pelagic
		Seasonal spawning migration of aquatic diadromous species, e.g., salmon, and nutrient input to terrestrial and freshwater ecosystems (Ben-David et al. 1998)	Forests, riparian vegetation, river and stream systems, estuaries
	Diffuse	Seasonal spawning migration of animals, e.g., land crabs, to the sea (Hicks 1985, Green et al. 2008) and predators, e.g., terrestrial mammals, feeding in intertidal systems (Carlton and Hodder 2003)	Forests, coastal vegetation, marine intertidal and subtidal
		Island systems subsidized by energy and nutrient inputs through highly mobile animals, e.g., seabirds and pinnipeds (Croll et al. 2005, Lombard et al. 2007, Sanchez-Pinero and Polis 2000)	Roosting, nesting and haul-outs islands, marine pelagic (feeding and high productivity areas)
		Movements of animals, e.g., seabirds between breeding (terrestrial) and feeding (marine) areas (Burger et al. 2000, Becker and Beissinger 2006)	Forests, marine pelagic
INTERFACES	*Narrow*	Input of marine nutrients into terrestrial systems by marine animals, e.g., marine iguanas and sea turtles (Wikelski and Trillmich 1994, Okey et al. 2004, Hannan et al. 2007)	Sand beaches, rocky shores, reefs, marine intertidal and subtidal
		Energy and nutrient inputs via terrestrial animals feeding in the intertidal zone, e.g., invertebrates and reptiles, into low productivity terrestrial ecosystems (Catenazzi and Donnelly 2007)	Coastal vegetation, dunes, intertidal
		Terrestrial-freshwater interface systems, e.g., stream systems and riparian vegetation, that capture sediments, nutrients, and pollutants (Reddy et al. 1999, Mulholland et al. 2008)	Forests, riparian vegetation, river and stream systems
	Broad	Systems, e.g., mangroves, that sustain species aggregations or provide protection for vulnerable life stages of terrestrial, freshwater, and marine species (Nagelkerken et al. 2008)	Forests, river and stream systems, mangroves, estuaries, reefs
		Freshwater-marine interface systems, e.g., seagrass, that trap sediments and organic matter and prevent erosion (de Boer 2007, Hutchings et al. 2005)	Mangroves, seagrass meadows, marine nearshore
		Terrestrial-freshwater interface systems, e.g., floodplains, that capture sediments, organic matter, and nutrients that can enter marine systems (Brunet et al. 1994, Seitzinger et al. 2006)	Terrestrial vegetation, floodplains

Table 2.1. Examples of ecological processes linking land and sea

Despite the abundant scientific literature on ecological linkages between realms, our understanding of the spatial and temporal dynamics of many land-sea processes remains limited. Importantly, spatially explicit data to represent these processes in planning are sparse. Examples of land-sea processes that, although incompletely understood, have been incorporated into planning include oceanic foraging by animals breeding on islands and coastal forests (e.g., Hazlitt et al. 2010; Lombard et al. 2007), salmon runs delivering nutrients to freshwater and terrestrial ecosystems (Ardron et al. 2002), riverine corridors important for downstream nutrient flows, and processes occurring in land-sea interfaces (Lagabrielle et al. 2009). The limited data on these interactions reflect a poor spatial understanding of processes generally (Pressey et al. 2007). Consequently, most planning exercises have either neglected or only incidentally incorporated these processes. Cost-effective methods to improve data and models to represent land-sea processes remain a major challenge for planners and scientists (Green et al. 2009).

Cross-system threats

Anthropogenic changes to the land or the sea can disrupt land-sea processes. In conservation planning generally, the crucial role of conservation areas is to mitigate or prevent proximate threats (Wilson et al. 2005a). Land-sea planning implies consideration of cross-system threats (Tallis et al. 2008) that can have significant impacts on coastal and marine biodiversity (Halpern et al. 2009; Stoms et al. 2005). Incorporation of cross-system threats involves identifying those that are most critical for coastal and marine conservation and mapping their sources and zones of influence, as well as assessing their magnitude and potential impacts (Allison et al. 1998; Wilson et al. 2005a). Previous studies have described and modelled direct threats originating in one realm and affecting others (see Abell et al. 2007; Cicin-Sain & Belfiore 2005; Halpern et al. 2008; Suski & Cooke 2007), but no synthesis of threats relevant to land-sea conservation planning exists.

Using the threat classification proposed by Salafsky *et al.* (2008), I identified 12 major cross-system threats and summarised their key characteristics: sources, affected realms, directions of influence, stressors, and areas to target for intervention (**Figure 2.1**). Important cross-system threats include: nutrient runoff from agriculture, which can

cause severe eutrophication, toxic phytoplankton blooms, and hypoxia in coastal ecosystems (Cloern 1996; Diaz & Rosenberg 2008; Howarth 2008); expansion of agriculture, forestry, and urbanisation, which lead to soil loss and increased sediment loads in rivers (Croke & Hairsine 2006; Walling 2006) and alter the structure and function of estuarine and coastal ecosystems (Thrush et al. 2004); and the use of pesticides in agriculture and forestry plantations, which results in dieback in mangroves and coral bleaching (Duke et al. 2005; Haynes et al. 2007; Hutchings et al. 2005).

Figure 2.1. Cross-system threats: sources, affected realms, direction, stressors and areas to target for action: **a)** Threat classes are based on Salafsky *et al.* (2008); **b)** On the basis of existing studies (Abell et al. 2007; Halpern et al. 2009; Halpern et al. 2007; Halpern et al. 2008) I anticipate that cross-system threats will have an array of impacts across the land-sea continuum, but a comprehensive review of these is not yet available. I suggest an initial broad scale of potential exposure to stressors and intensity of impacts associated with cross-system threats (see Wilson et al. 2005a); **c)** a classification commonly used identifies point and nonpoint sources of pollution, which can be used to classify sources of cross-system threats. Point sources can be addressed directly by managing individual facilities or structures, whereas nonpoint sources require actions associated with areas or regions; **d)** threat agents or stressors are defined as environmental and biotic factors that exceed their natural ranges of variation owing to human activities (Mullan-Crain et al. 2008).

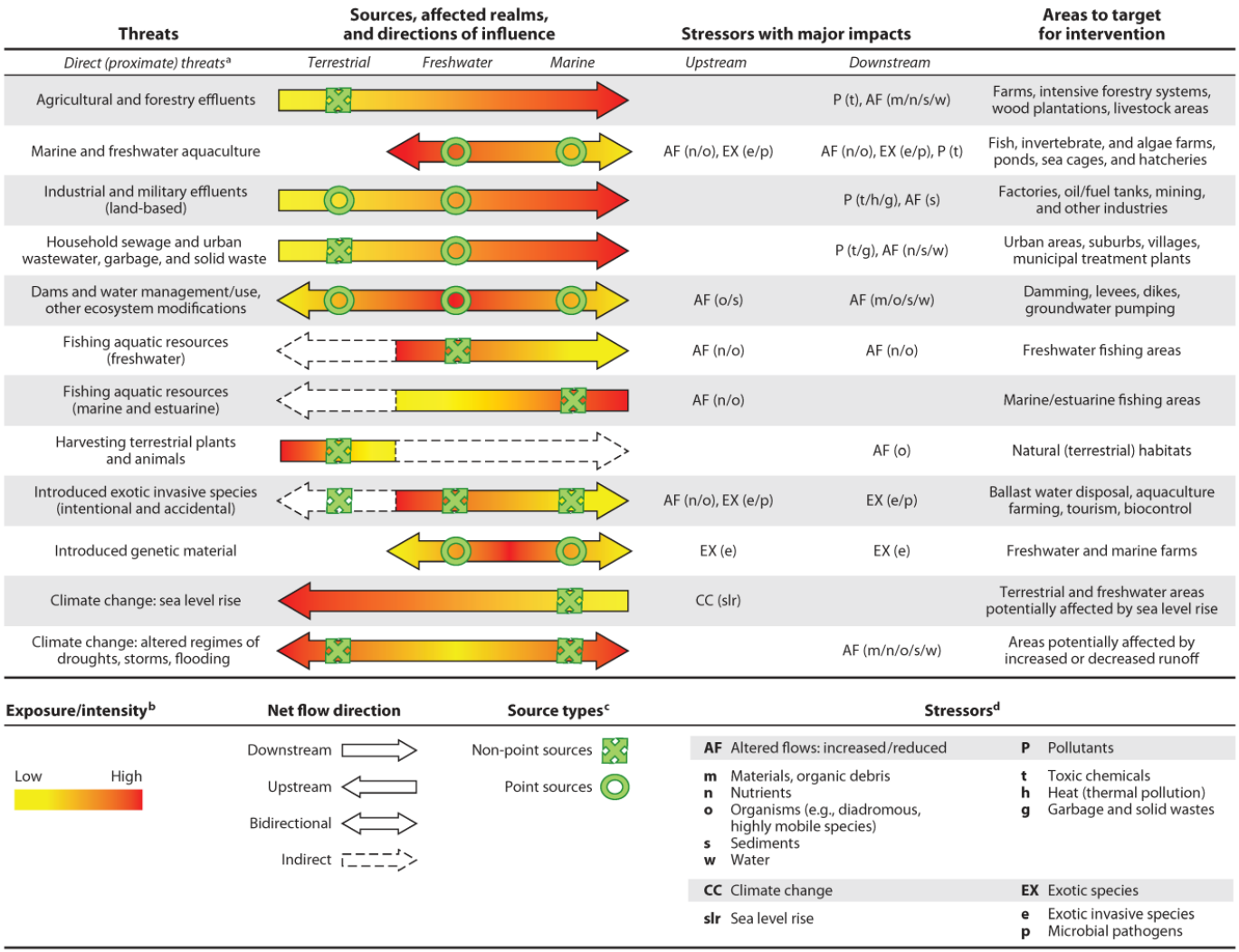

Figure 2.1. Cross-system threats: sources, affected realms, direction, stressors and areas to target for action

Cross-system threats most commonly originate on land and affect the sea, mediated by rivers (Stoms et al. 2005), but terrestrial and freshwater systems also can be affected by activities in the marine realm. In addition to direct impacts on salmon spawning, fishing in marine and estuarine areas, for instance, can significantly reduce marine-derived nutrients delivered to rivers (Gresh et al. 2000) and potentially affect populations of terrestrial predators feeding on salmon as well as the delivery of nutrients to riparian forests (Helfield & Naiman 2002, 2006). Some threats can have both downstream and upstream impacts, such as damming and other water management practices. Dams can alter the flows of sediments, nutrients, and water, resulting in downstream changes to coastal ecosystems (e.g., changes in phytoplankton species composition: Humborg et al. 2000) and decreased productivity of fishing areas (e.g., as reported for Cuban fisheries: Baisre & Arboleya 2006), but they also affect upstream migration and spawning of anadromous fish, thus potentially also reducing productivity of freshwater and riparian ecosystems (Naiman et al. 2002). Cross-system threats affecting diffuse connections (**Table 2.1**) can be difficult to identify and measure. For example, reduced reproductive success of the endangered marbled murrelet (*Brachyramphus marmoratus*), a seabird that depends on old-growth coastal forests to breed, has been related to overexploitation of the coastal fish stocks on which it feeds (Becker & Beissinger 2006). In this case, both forest protection and fisheries management are necessary components of a successful conservation plan (Burger et al. 2000; Hazlitt et al. 2010).

An assessment of both single-realm and cross-system threats is needed to determine suitable locations for conservation areas (Stoms et al. 2005; Wilson et al. 2005a). In general, including highly vulnerable areas in conservation networks will compromise conservation objectives if the probability of persistence of their valued features is low (Beck 2003; Game et al. 2008; Wilson et al. 2005a), especially if conservation of alternative areas can achieve the same objectives. For this reason, planners often aim to avoid imminent threats where possible. Potential replacements for more threatened areas can be identified in several ways, including analysis of irreplaceability (Margules & Pressey 2000), which can provide maps of options for achieving objectives and give scope for participation and negotiation. Depending on circumstances, strategies other than avoidance of current or potential threat might be appropriate, especially if highly irreplaceable marine areas fall within the zone of influence of cross-system threats. **Figure 2.2** illustrates a decision tree that can be used to evaluate potential conservation

areas exposed to cross-system threats. Strategies to manage cross-system threats will depend on irreplaceability, the intensity of threats, and the likely effectiveness and costs of abating them (Klein et al. 2010).

Figure 2.2. Example of a decision tree to evaluate potential conservation areas considering cross-system threats: Scenario 1) Potential conservation areas that are not affected by cross-system threats but contain features of conservation interest that are threatened locally can be considered for inclusion in a marine protected area (MPA) system and protected or managed accordingly. Scenario 2) Potential conservation areas that are affected by cross-system threats and have low irreplaceability values (numerous possible replacements) can be substituted by less affected or unaffected areas. Scenario 3) If irreplaceability is high (spatial options are limited) and threats can be managed or abated (e.g., by improving water quality through better agricultural and forestry practices to reduce nutrient runoff or excessive sedimentation), then actions to manage the source of threats (e.g., in coastal catchments) should be considered along with the local protection of the affected areas. Scenario 4) If cross-system threats cannot be abated, but the area does not include features considered exceptional (e.g., high irreplaceability might be due to one or more features that also occur extensively in neighbouring regions), then planners could still try to find replacements to achieve objectives related to features characteristic of the region. However, if a vulnerable area is highly irreplaceable for characteristic features, it could be included in the conservation area system (e.g., MPA network), although the viability of some of its features might be reduced. In these cases, the protection of the area could be used as a strategy to leverage management actions to reduce the impacts of cross-system threats (Scenario 5) and also to increase the resilience of the area to these threats (e.g., by restoring habitat or managing local threats, such as overexploitation). Alternatively, planners might exclude the area from consideration as part of the conservation network and acknowledge the potential loss of the associated biodiversity features (Scenario 6).

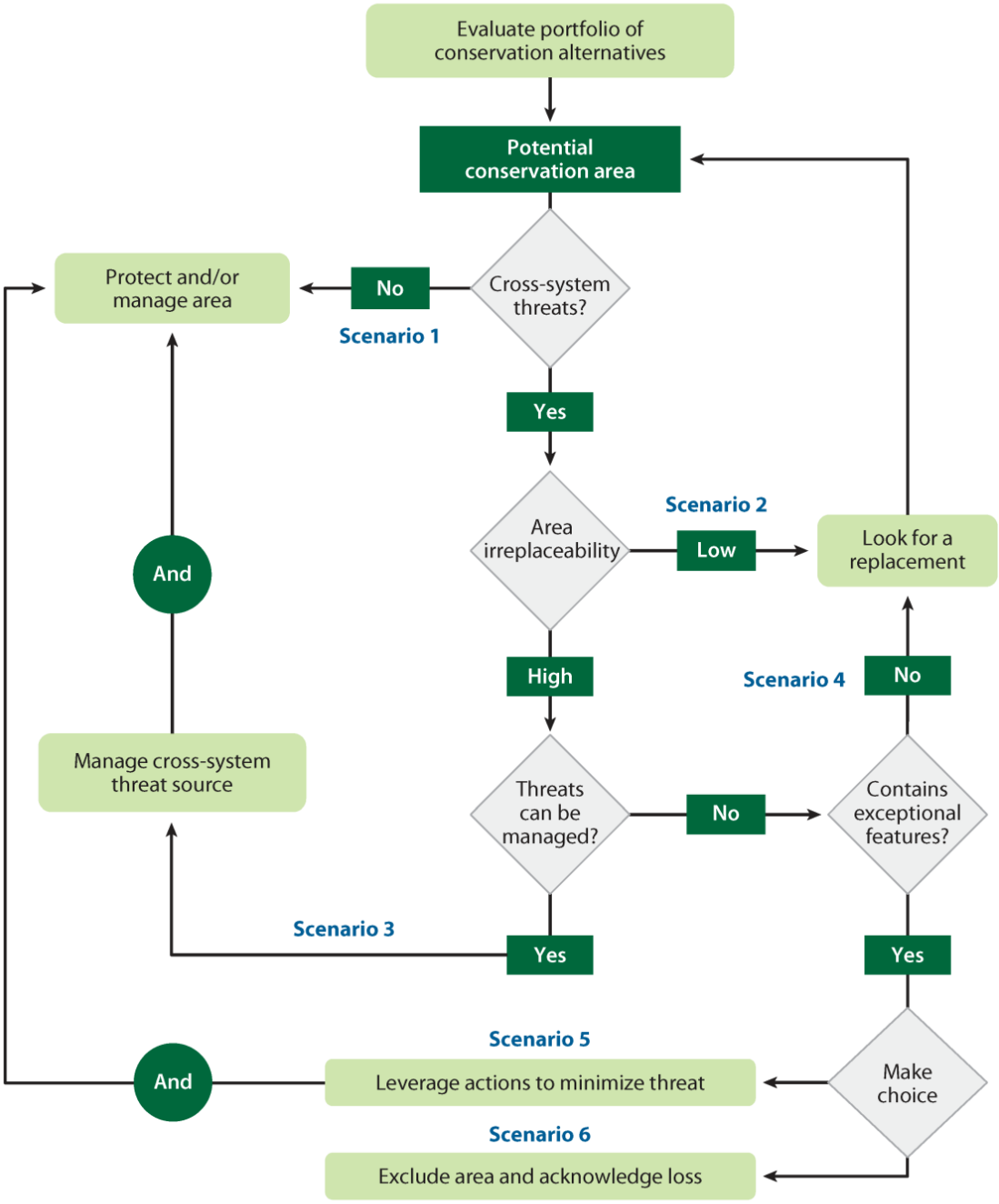

Figure 2.2. Example of a decision tree to evaluate potential conservation areas considering cross-system threats

Socioeconomic interactions

People are an important link between the land and sea, and hence solutions to land-sea planning should consider socioeconomic interactions between realms. Most of the worldôs population lives in the coastal zone and therefore strongly influences land-sea processes (Cicin-Sain & Belfiore 2005) and drives cross-system threats (**Figure 2.1**). Curtailing these threats therefore involves managing peopleôs use of the environment across the land-sea continuum, thereby affecting people positively and negatively on land (Carwardine et al. 2008) and in the sea (Klein et al. 2008b). Nonetheless, planning for the socioeconomic benefits and impacts of conservation actions has been undertaken only within single realms.

Conservation actions in one realm can have socioeconomic implications in another (Cruz-Trinidad et al. 2009), particularly in regions where land-sea processes and/or cross-system threats are strongly linked to local livelihoods. Consider the case of a forested catchment that drains into a near-shore coral reef ecosystem. If the coastal forest is converted to an intensive land use, such as oil palm production, increased pollutants could negatively affect the reef (Beger & Possingham 2008; Huber 1994) and potentially also damage marine tourism and fisheries. In the Chesapeake Bay, for example, eutrophication derived from agriculture has contributed to declines in demersal fish, resulting in a shift to fishery landings dominated by pelagic species (Kemp et al. 2005). Alternatively, if the coastal forest is protected and contains charismatic wildlife, tourists visiting the marine area might also visit the forest, thus providing additional income to the local economy. In addition to coordination, managing across realms requires social and economic foresight. For example, establishment of an MPA might force fishers to supplement their income by extending land-based activities such as forestry and agriculture, thereby negatively affecting terrestrial biodiversity.

Socioeconomic interactions between land and sea also involve feedbacks at various scales. Locally, displacement of fishers by an MPA leading to intensification of their land-based activities might result in higher sediment and nutrient discharges into the sea. This in turn could negatively affect the MPA, potentially undermining the values that motivated its establishment. Furthermore, pollutants and runoff might negatively

affect the remaining fishing grounds, causing more fishers to switch to forestry or agriculture, leading to further negative feedbacks. At broader scales, coral bleaching can be exacerbated by nutrient runoff from coastal catchments (Wooldridge 2009), negatively impacting regional economies through loss of tourism and fishing resources (Hoegh-Guldberg et al. 2007). Thus, the absence of comprehensive land-sea planning can lead to altered socioeconomic interactions between land and sea, often with negative consequences for one or both realms.

Advancing land-sea planning: an operational framework

Given the strong and diverse ecological and socioeconomic couplings between land and sea, how can conservation science inform more effective planning? I developed an operational framework to guide land-sea conservation planning (**Figure 2.3**) that is informed by expert opinion and case studies that considered land-sea connections. The framework emphasises the elements that are particularly relevant to land-sea planning and has a loose structural match with the eleven stages of Pressey & Bottrill (2009). I systematically selected 26 case studies (key aspects are summarised in **Supplementary Material**) that applied the general principles of systematic conservation planning, identified conservation objectives in both marine and terrestrial realms, and considered cross-system threats and/or land-sea processes. Although each study addressed several components of the framework in **Figure 2.3**, some components were missing from all studies. This indicates the need for further development of land-sea planning and that different components can be more or less important under different circumstances.

Figure 2.3. Operational framework for integrated land-sea planning. The framework is comprised of three broad phases: regional scoping, integration and analysis, and synthesis. External arrows represent flows of information and feedbacks in the adaptive planning process, and they highlight the need to understand land-sea planning as a continuous and updatable process. Arrows encircling the second phase indicate that planning involves iterative cycles of analysis in which conservation objectives, design criteria, and spatial data and models are adjusted to explore potential solutions under different environmental and socioeconomic scenarios.

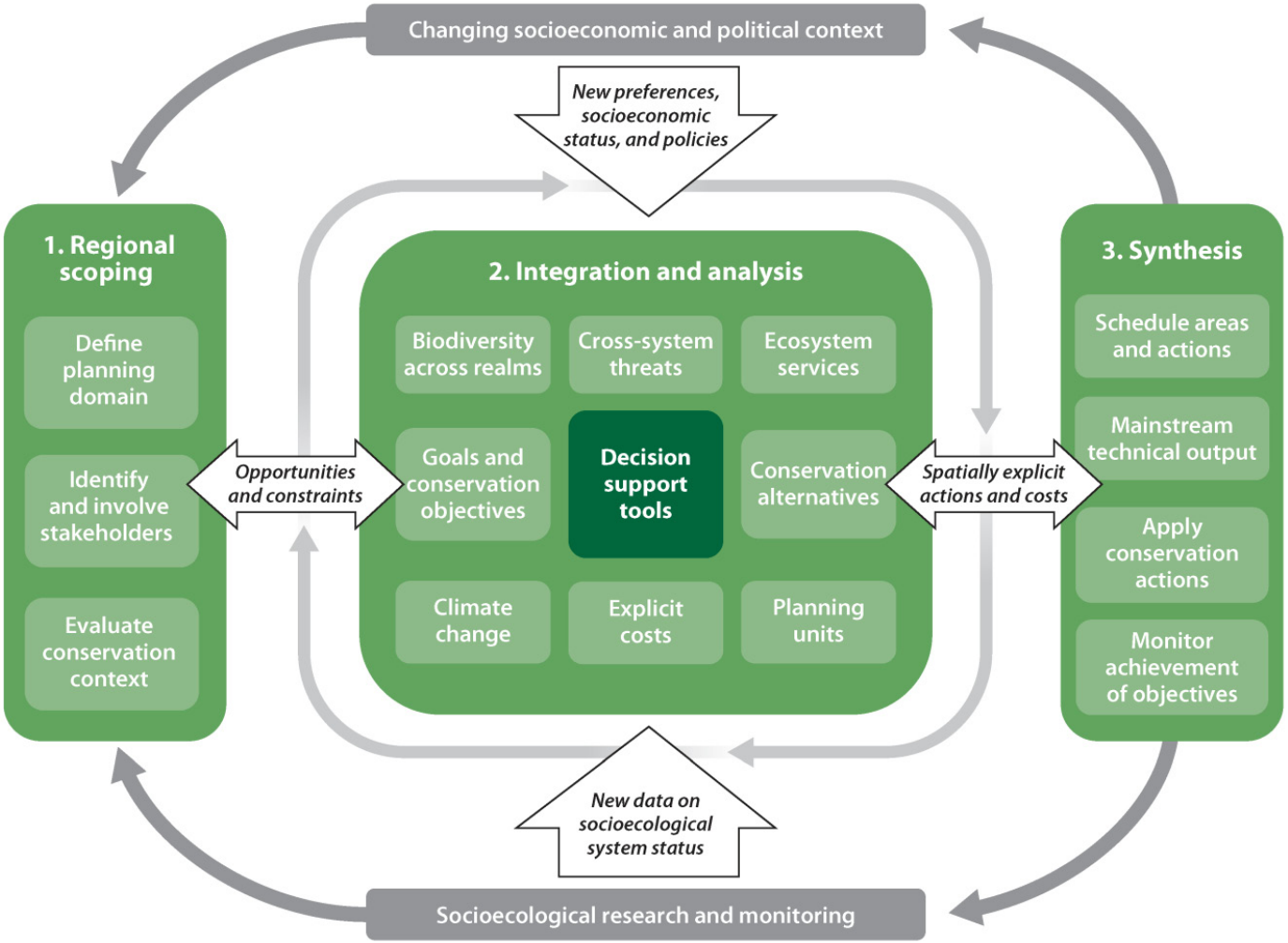

Figure 2.3. Operational framework for integrated land-sea planning

The framework has three broad phases: regional scoping, integration and analysis, and synthesis. It includes familiar components of conservation planning adapted to a land-sea approach and some less generic aspects particularly important for land-sea planning. Explicit in the framework is an adaptive approach (Walters & Hilborn 1978) that is indicated by feedbacks that link the phases and reflect the iterative nature of planning. Feedbacks requiring reconsideration of earlier parts of the process might follow, for example, anthropogenic changes to land/seascapes, altered social and economic conditions, new data sets, and monitoring of the effectiveness of conservation actions (Sarkar et al. 2006). An adaptive approach is particularly relevant for land-sea planning owing to the uncertainty and complexity associated with planning across realms. Among the sources of uncertainty are our limited knowledge of land-sea interactions, lag times between implementation of actions and measurable results, and difficulties in predicting climate change, land-use change, and their potential impacts on water quality and marine ecosystems (Broderick 2008). The phases of the framework and their components are described in the following sections.

Regional scoping

Land-sea planning involves several additional considerations about aspects of scale and regional scoping. Scoping components lay the foundation for a better understanding of the broad opportunities and constraints that will influence the implementation of conservation actions and set the scene for the following planning phases. Although these issues are critical to conservation planning success, they are less tied to the ecological issues at the core of this review and are not examined in detail here.

Defining the planning domain

An integrated land-sea approach requires identification of the sources and zones of influence of cross-system threats and spatial representations of land-sea processes (from upland areas to the nearshore and sometimes offshore marine environment), both challenging tasks.

Defining the limits of the planning domain in the terrestrial realm is not always straightforward because land that influences marine areas can extend well inland. For

example, waters that discharge into the Gulf of Mexico originate from an area of almost 3 million km2, or approximately 40% of the United States (Mitsch et al. 2001). A single planning process at this scale is extremely difficult because of the diversity of jurisdictions and stakeholders that would need to be involved as well as the time, expense, and requirements to collect consistent data. Full-catchment approaches have covered small catchments (e.g., the 425 km2 catchment of Jervis Bay, Australia, Dutton et al. 1994) or larger and multiple catchments, albeit at coarse resolution (e.g., all coastal catchments draining into the Mesoamerican Caribbean Reef, Kramer & Kramer 2002). Other planning exercises have focused on the downstream ends of major rivers draining into the ocean (e.g., Chesapeake Bay Lowlands Ecoregional Plan, TNC 2002) or portions of catchments with major direct impacts on marine ecosystems that are likely to be developed in the near future (e.g., Mission Aransas Estuary, Texas, Dunton 2009).

Stakeholders and conservation context

Identifying and involving stakeholders (Pomeroy & Douvere 2008) and evaluating the conservation context (Pressey & Bottrill 2009) involve critical social and political analyses that bring diverse views and goals into discussion and place the subsequent spatial analyses in the context of multiple social, economic, and political factors operating across terrestrial and marine realms (Cicin-Sain & Belfiore 2005).

Integration and analysis

The second phase integrates biological and socioeconomic information, which involves adapting several familiar components of conservation planning to a land-sea approach (e.g., planning goals and objectives, and socioeconomic data and models). Other less typical aspects include incorporating land-sea interactions and adapting decision support systems to address multiple objectives that include land-sea interactions.

Goals and conservation objectives

Goals, framed qualitatively, provide the link between values and the quantitative analyses that characterise systematic conservation planning (Pressey & Bottrill 2009).

In addition to goals associated with individual realms (e.g., protection of terrestrial and marine species and habitats), goals for land-sea planning include maintenance and restoration of land/seascape connectivity - defined by natural flows of organisms, water, materials, and energy - to ensure the persistence of biodiversity across realms (Bennett et al. 2006; Talley et al. 2006). Broad goals to maintain marine ecological processes (see Banks et al. 1999; Kramer & Kramer 2002) ideally should be expanded into more explicit statements about specific land-sea linkages, such as maintaining nutrient flows between marine and terrestrial ecosystems (see Lagabrielle et al. 2009; Lombard et al. 2007). Goals referring to mitigation of cross-system threats also should be specific. Some examples are the identification of areas or catchments to be protected or restored to minimise downstream impacts on marine ecosystems and fisheries in Fiji (Atherton et al. 2005) and the Coral Triangle (Klein et al. 2010). Goals related to integrated land-sea conservation have been rare, or stated but not translated into strategies to achieve them. Reasons include limited resources, lack of information on land-sea interactions, differences in stakeholdersô interests, and stakeholdersô lack of awareness of the importance of these linkages.

Once stakeholders agree upon goals, they need to be translated into (preferably) quantitative conservation objectives (Pressey & Bottrill 2009). Land-sea planning objectives should include those associated with spatial features that represent land-sea processes and areas to be managed or protected to mitigate cross-system threats. Planning exercises have identified diverse features associated with maintaining land-sea processes. For example, Ardron et al. (2002), in designing a network of MPAs for the central coast of British Columbia, mapped and targeted streams that support migrations of anadromous species. Other studies have targeted keystone species that deliver marine-derived nutrients to terrestrial ecosystems (e.g., seabirds and pinnipeds of the Prince Edward Islands, South Africa, Lombard et al. 2007) and freshwater ecosystems (e.g., salmon and steelhead in the Alaska Peninsula and Bristol Bay Basin, TNC 2004). Others have focused on species with life cycles reliant on both terrestrial and marine habitats, such as the land crab, which migrates to coastal waters to spawn (MPO 2003), and the whooping crane, which moves between grasslands and salt marsh (Dunton 2009).

Conservation objectives related to land-sea interfaces or linking ecosystems (e.g., estuaries, mangroves, coastal streams) or their associated species (e.g., crocodiles, estuarine turtles and fish, waterbirds) can also contribute to land-sea integration (e.g., Florida Marine-Estuarine Site Assessment, Geselbracht et al. 2005; Geselbracht et al. 2009).

Cross-system threats can be minimised by targeting potential sources of detrimental flows (e.g., areas with a high erosion potential, Atherton et al. 2005), preferably in combination with setting quantitative water quality objectives (e.g., reduction of annual pollutant discharges from catchments in kilograms of nitrogen per hectare per year). Water quality objectives can be based on historic and current discharges as estimated by catchment modelling and monitoring, and ideally on assessment of exposure thresholds or tolerance of marine species or habitats to pollutants as well (Brodie et al. 2009).

Biodiversity across realms

Land-sea planning requires spatially explicit data on biodiversity patterns and ecological processes across the land-sea continuum. These include distribution models or occurrences of marine, terrestrial, and freshwater species, communities, and habitat types as well as maps of features associated with land-sea processes. Data and models are commonly focused on a single realm or separate realms, although these might include some species occurring in more than one realm (e.g., diadromous and migratory species of economic and ecological importance, TNC 2003; Vander Schaaf et al. 2006). The types and resolutions of spatial data on biodiversity vary across realms but are generally more diverse and detailed in terrestrial environments (Avery 2003).

Collection of data on and mapping of land-sea processes requires identification of the features that will serve as surrogates to be represented in conservation areas (Cowling et al. 1999). Building on studies that explicitly integrate processes into planning (Cowling & Pressey 2003; Cowling et al. 1999; Rouget et al. 2003a) and methods to incorporate processes across realms proposed by Beger et al. (2010a), I identified four potential strategies to plan for land-sea processes: design criteria, variable objectives, planning units, and movable conservation areas (**Table 2.2**). Few studies have addressed the design needs of land-sea processes in a spatially explicit way. Lagabrielle et al. (2009)

illustrated for Reunion Island the need to create an inland buffer along the coastline to protect some land-sea processes, including settlement of new species and movements of organisms between land and sea. As part of the design of the Prince Edward Islands MPA, Lombard et al. (2007) included portions of oceanic foraging areas based on the major movement axes of seabirds and pinnipeds to maintain the input of marine-derived nutrients into terrestrial systems. In British Columbia, Ardron et al. (2002) incorporated the flow of nutrients between oceans, streams, and riparian areas by setting objectives for key salmon-spawning streams, estuaries, and holding sites. Protection of extensive processes, such as the bidirectional nutrient flows mediated by the migration of salmon, could also require the protection of terrestrial features (e.g., riparian vegetation, upland forests) and management of freshwater networks that are currently overlooked in marine planning, but this is yet to be demonstrated.

Table 2. Strategies to incorporate land-sea processes into conservation planning

Design criteria	**Spacing:** Define a minimum and/or maximum distance between potential conservation areas that contain complementary terrestrial/freshwater and marine habitats to ensure biological or physical connectivity between realms or to reduce the risk of disturbances affecting multiple conservation areas simultaneously.
	Buffers: Define buffers for conservation areas along terrestrial-marine interfaces or around other areas that support land-sea processes. The roles of buffers are to ensure that areas supporting the processes of interest are managed appropriately and that, as far as possible, threatening processes are excluded.
	Replication: Represent terrestrial or marine features associated with land-sea processes in multiple conservation areas to minimize simultaneous exposure to natural or anthropogenic disturbances.
	Adjacency: Place terrestrial/freshwater and marine conservation areas beside each other to maintain connectivity and promote the persistence of land-sea processes.
	Alignment: Include the major movement axes of organisms (e.g., between the foraging and nesting areas of seabirds) or the locations of physical features (e.g., ocean fronts) associated with land-sea processes.
	Connectivity: Select areas for conservation management that promote one or more relevant aspects of connectivity. These could include continuous corridors of suitable aquatic or terrestrial areas, minimum spacing between areas to facilitate movement of organisms, or simultaneous selection of areas that are functionally, even if not spatially, connected (e.g., to allow for organisms inhabiting different realms to move between them to complete their life cycles or to recolonize areas through the movement/transport of planktonic larvae, considering currents and larval biological characteristics and behaviour).
Variable objectives	**Habitats:** Set objectives for both terrestrial/freshwater and marine habitats that are linked through movements of organisms (e.g., wetlands, estuaries, coral reefs) and that need to be protected simultaneously, perhaps with maximum allowable separation distances, to sustain these interactions.
	Species: For organisms that periodically move between realms or that have life stages associated with different realms, set separate objectives for terrestrial/freshwater and marine occurrences to protect their complete life cycles.
	Stratification: To fully represent the internal heterogeneity of narrow land-sea interfaces, a general (larger) objective can be assigned to the interface type of interest as a whole (e.g., 20% of rocky intertidal shores) and specific (progressively smaller) objectives to lower classes or sub-categories identified at a finer scale/resolution (e.g., 10% and 5% of exposed and lower exposed rocky intertidal shores, respectively).
Planning units	**Special:** Delineate planning units (e.g., linear, irregular) that are configured specifically to represent interfaces associated with land-sea processes of interest.
	Smaller: Use smaller planning units in interface habitats to recognize the greater spatial heterogeneity found in and around these areas as well as to increase selection precision and minimize over representation.
Movable areas	**Spatial:** Apply conservation actions to different areas at different times according to the occurrences of dynamic physical features and associated species' habitats.
	Temporal: Apply conservation actions in the same areas at different times or seasons to protect vulnerable life stages of species moving across realms (e.g., during migration, reproduction, or spawning) or flows (e.g., peak river discharges) that sustain land-sea processes.

Table 2.2. Strategies to incorporate land-sea processes into conservation planning

Cross-system threats

Understanding the dynamics and potential impacts of cross-system threats is necessary to develop spatial alternatives for conservation areas that can minimise vulnerability and maximise conservation outcomes. Models of cross-system threats can provide planners with four kinds of information: sources, zones of influence, potential impacts, and expected changes in intensity or patterns in response to land/seascape dynamics or to conservation actions. Although planning exercises have modelled and incorporated cross-system threats (e.g., Crist et al. 2009; Green et al. 2009; Tallis et al. 2008), none has addressed them comprehensively through the four types of information above.

Identifying actions to effectively mitigate cross-system threats depends on mapping and modelling of those threats, which can be demanding of time and data. Spatial precision might be infeasible in some circumstances, so coarse-resolution models to identify general patterns at regional scales might be necessary. Models of threats are usually land-based and have generally neglected threats originating in marine and freshwater systems (**Figure 2.1**). Potential zones of influence of point sources are usually mapped as single or multiple-ringed buffers associated with different levels of impact (Green et al. 2009; Ulloa et al. 2006), but they also can be modelled in more complex ways if directional data on intensity and effects on ecosystems are available. This might be guided by expert opinion (Halpern et al. 2009; Klein et al. 2010). Mapping nonpoint sources involves estimating the potential contribution of different areas to the extent or intensity of threats (e.g., pollutant discharge), usually through catchment models (Crist et al. 2009; McKergow et al. 2005b), followed by mapping the zone of marine influence using diffusion models (Dunton 2009; Halpern et al. 2008; TNC 2008) or river plumes models (Cherubin et al. 2008; Paris & Cherubin 2008; Tallis et al. 2008).

Ecosystem services

Ecosystem services describe what people value from ecosystems and have become prominent components of planning exercises (Chan et al. 2006; Egoh et al. 2007). The importance of land-sea processes in the provision of diverse ecosystem services (e.g., pollution buffering and sediment trapping is well recognised, Granek et al. 2010; Silvestri & Kershaw 2010), but these generally have been overlooked in setting explicit

conservation objectives. Some land-sea processes are critical to sustaining coastal and marine ecosystems and species of economic importance or have been linked to other important processes, such as maintenance of oceanic and terrestrial productivity (**Table 2.1**). Maintaining the services provided by coastal and marine ecosystems (e.g., nutrient cycling) and the economic benefits associated with these (e.g., for fisheries and tourism) are commonly recognised as broad planning goals (Beck & Odaya 2001; Green et al. 2009; Hinchley et al. 2007), but seldom have been translated into spatial data and quantitative conservation objectives. A study conducted by The Nature Conservancy in Floridaôs Northwest Coast (TNC 2007) exemplifies a method to incorporate ecosystem services provided by coastal wetlands, in this case hazard mitigation, into conservation planning.

Most studies have not considered ecosystem services or have limited their attention to provisioning services such as sustaining fisheries. Some services supported by land-sea processes might be incorporated incidentally through objectives for other features, but others will remain unprotected, and their persistence will therefore remain at risk. Silvestri and Kershaw (2010) developed a framework to help planners understand, map, and value ecosystem services along the land-sea continuum and to determine the spatial alternatives for their protection or restoration.

Climate change

Of special concern for land-sea planning are the effects of climate change on land-sea processes and cross-system threats. Climate change--associated alterations in land-sea processes, such as migration of fish and nutrient runoff, are anticipated (Nohara et al. 2006; Reist et al. 2006). Projected increases in temperature and altered precipitation regimes will likely result in more intense nutrient and sediment runoff associated with land use change, at least in some regions within the tropics (Nohara et al. 2006; Zhu et al. 2008). Synergistic effects of hypoxia, temperature, and CO_2 levels on the physiology of marine organisms are also expected to intensify with climate change (Harley et al. 2006). The projected sea level rise will probably affect land-sea interface ecosystems (Gilman et al. 2008) as well as terrestrial communities by reducing habitat for species in coastal zones . Accordingly, impacts of climate change on biodiversity and threats, and the interactions between them, should be considered for both land and sea.

Although climate change is mentioned as a major concern in most planning exercises, few have explicitly modelled climate-related changes, and most of these are restricted to species and habitats associated with a single realm or separate realms (e.g., shifts in climatic variables across terrestrial ecosystem units to identify potential refuges, TNC 2005). More often, general strategies or design principles are followed with the aim of increasing the resilience of ecosystems to climate change by protecting key habitats and ecosystems. For example, the MPA network designed for Kimbe Bay, Papua New guinea, considered spreading risks through representation and replication, protection of special sites (e.g., spawning aggregations, nesting areas, and sites more resistant to climate change), connectivity (maximum separation of 15 km), and minimum MPA size (10 km^2) (Green et al. 2009). Similar efforts guided MPA network design along the California coast under the Marine Life Protection Act (Gleason et al. 2010).

Explicit costs

Integration of socioeconomic considerations in land-sea planning is challenging but can be attempted by minimising the costs of conservation actions to people (Naidoo et al. 2006). A common approach in single-realm planning is to minimise costs of management actions while achieving conservation objectives (Carwardine et al. 2008). In land-sea planning, costs are commonly integrated into cost-suitability indices developed for terrestrial and marine realms independently. Cost-suitability indices integrate socioeconomic, political and biological factors to preferentially select areas with higher long-term viability and avoid areas with high conservation costs (Geselbracht et al. 2009; TNC 2005). Ideally, land-sea planning should consider costs in both realms simultaneously. For example, a study in the coral triangle region, where coral reef conservation was the main goal, considered the costs of conservation actions on the land (i.e., costs associated with management of terrestrial protected areas and forgone economic return from cropping and grazing to reduce land-based threats) and in the sea (i.e., MPA management costs and associated opportunity costs for fishermen) (Klein et al. 2010). Although the costs were not expressed in economic terms, Tallis et al. (Tallis et al. 2008)considered costs for marine conservation associated with land-based threats. There might, however, be trade-offs between minimising cross-system threats and minimising negative socioeconomic impacts across the land-sea interface.

For example, if much land needs to be protected to ensure the persistence of marine biodiversity potentially affected by land-based pollution, but people rely on agriculture for subsistence, the cost of terrestrial protected areas could be unfeasibly high. In these cases, other conservation actions, such as better agricultural practices, may minimise downstream impacts at lower cost (Gordon 2007).

Planning units

The delineation of planning units in land-sea planning is influenced by the same factors considered in terrestrial and marine exercises, including data resolution, size of planning region, and socioeconomic and political factors linked to implementation (Cowling & Pressey 2003; Pressey & Logan 1998). In land-sea planning, another consideration for the choice of planning units is the need to integrate conservation areas across the land-sea interface, which often requires different units for the terrestrial (e.g., subcatchments, ownership parcels), marine (e.g., hexagons), and interface (e.g., linear) parts of the planning domain. Small polygonal units (Geselbracht et al. 2005) can also be useful to recognise the spatial heterogeneity found in narrow interfaces, to increase selection precision, and to minimise overrepresentation (see **Table 2.2** and Beger et al. 2010). Banks et al. (2005) used linear units to configure a system of representative intertidal conservation areas on the Queensland (Australia) coast. A study in the U.S. Pacific Northwest used diverse units: terrestrial and freshwater features were summarised in subcatchments; estuaries were each mapped as single planning units; and square units intersected the coastline, offshore islands, and upstream tidal reaches of major rivers (Vander Schaaf et al. 2006). In contrast, a planning exercise in Alaskaôs Cook Inlet used a uniform hexagonal grid that covered the land and sea portions of the planning domain (TNC 2003). Until now, most planning exercises have delineated planning units independently for the land and the sea, and their design and configuration have not accounted for linkages between the realms.

Decision support tools

Major challenges to land-sea planning come with the need to manage data from terrestrial, freshwater, and marine ecosystems, to consider connectivity between realms, and to relate potential conservation areas to sources and impacts of cross-system threats.

Decision support tools are therefore key elements in land-sea planning. Incorporation of connectivity between realms can require modification of data or adaptation of existing tools and algorithms (beger *et al.* 2010, mumby 2006, stoms *et al.* 2005). Beger *et al.* (2010) examined existing and new methods to incorporate ecological linkages between realms supported by conservation planning software, but few of these approaches have actually been used. Approaches to integration in decision-support systems include design criteria to force the selection of marine conservation areas close to well-protected terrestrial or estuarine areas and away from potential cross-system threats (fernandes *et al.* 2005, green *et al.* 2009, tallis *et al.* 2008). They also include simultaneous prioritisation across realms (tnc 2003, 2005) and evaluation of current and projected land-use scenarios to explore alternatives for mitigation of threats (dunton 2009). Land-sea planning will require the use and adaptation of existing conservation planning software interfaced with other modelling and simulation tools to integrate the diversity of data and models required (e.g., crist *et al.* 2009, tnc 2007).

Conservation planners often use numerical optimisation tools (e.g., C-Plan, Marxan, Zonation) to help identify priorities for conservation. Two studies demonstrate methods for integrated land-sea planning using Marxan (ball *et al.* 2010). Tallis *et al.* (2008) explored changes in the distribution of threats and the configuration of marine priority areas when cross-system threats were explicitly considered. They incorporated the impact of river-derived threats to identify conservation priorities in the u.s. Pacific northwest (**Figure 2.4**). Hazlitt *et al.* (2010) incorporated a land-sea process to identify priorities for conservation of the marbled murrelet (a species that requires old-growth forest for nesting and high-quality marine habitats for foraging) in British Columbia, Canada. For this species, they found that the inclusion of marine objectives (a function of quality of and distance to potential suitable marine foraging habitats) in the planning process influenced the location of priority areas for terrestrial reserves (**Figure 2.5**), especially when conservation resources only allowed for the protection of a small fraction of available terrestrial habitat.

Figure 2.4. Differences in the spatial location of marine conservation priorities between scenarios that incorporated and ignored cross- system threats. A) Only single-system threats considered; B) planning units within the plume of the Columbia River (enclosed by the white line) had higher costs for marine conservation. Most coastal marine areas within the zone of influence of the river plume were not selected. Copyright É 2008 Wiley. Figure reproduced and modified from Tallis *et al.* (2008) with permission from the authors.

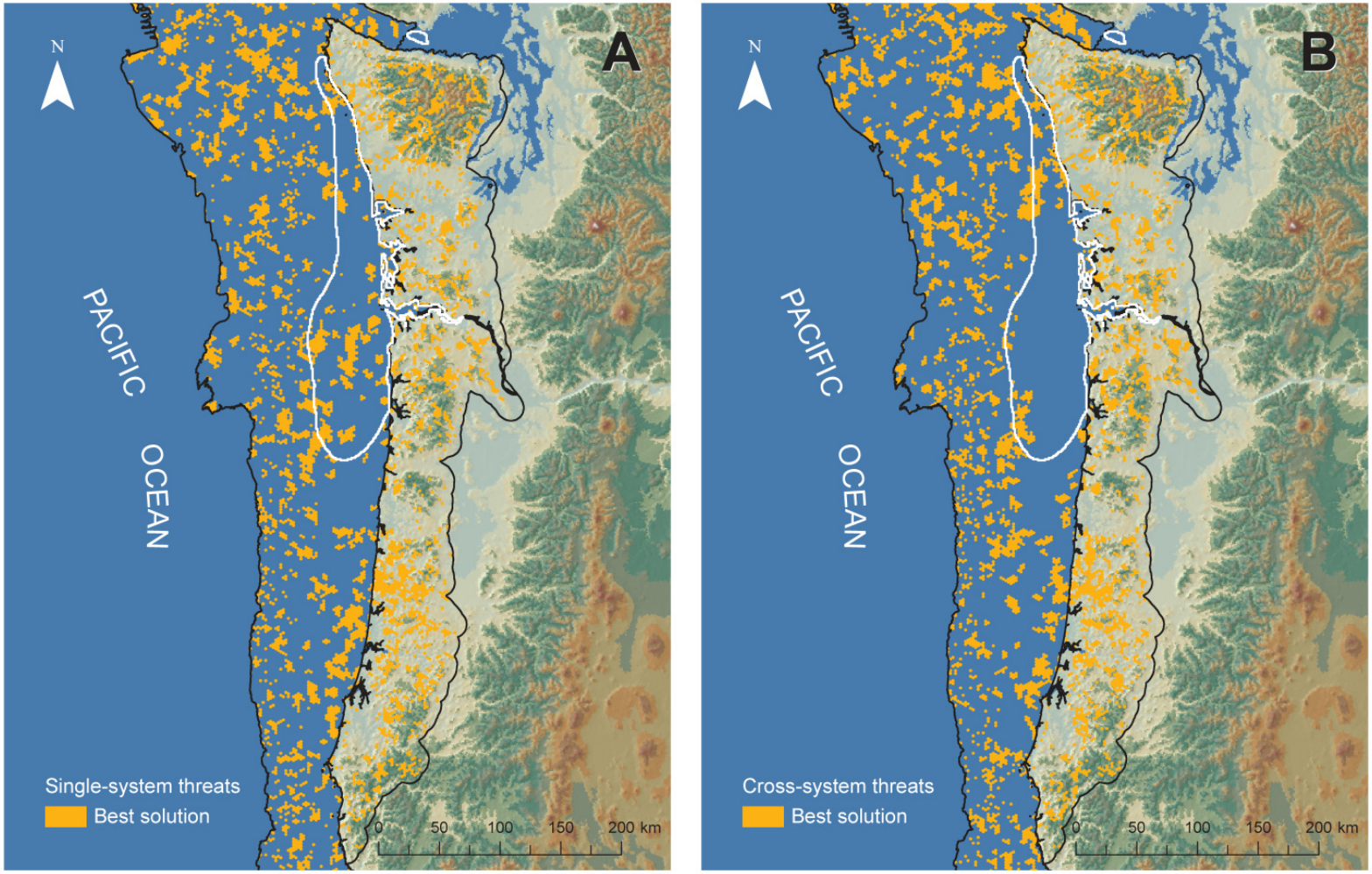

Figure 2.4. Differences in the spatial location of marine conservation priorities between scenarios that incorporated and ignored cross- system threats

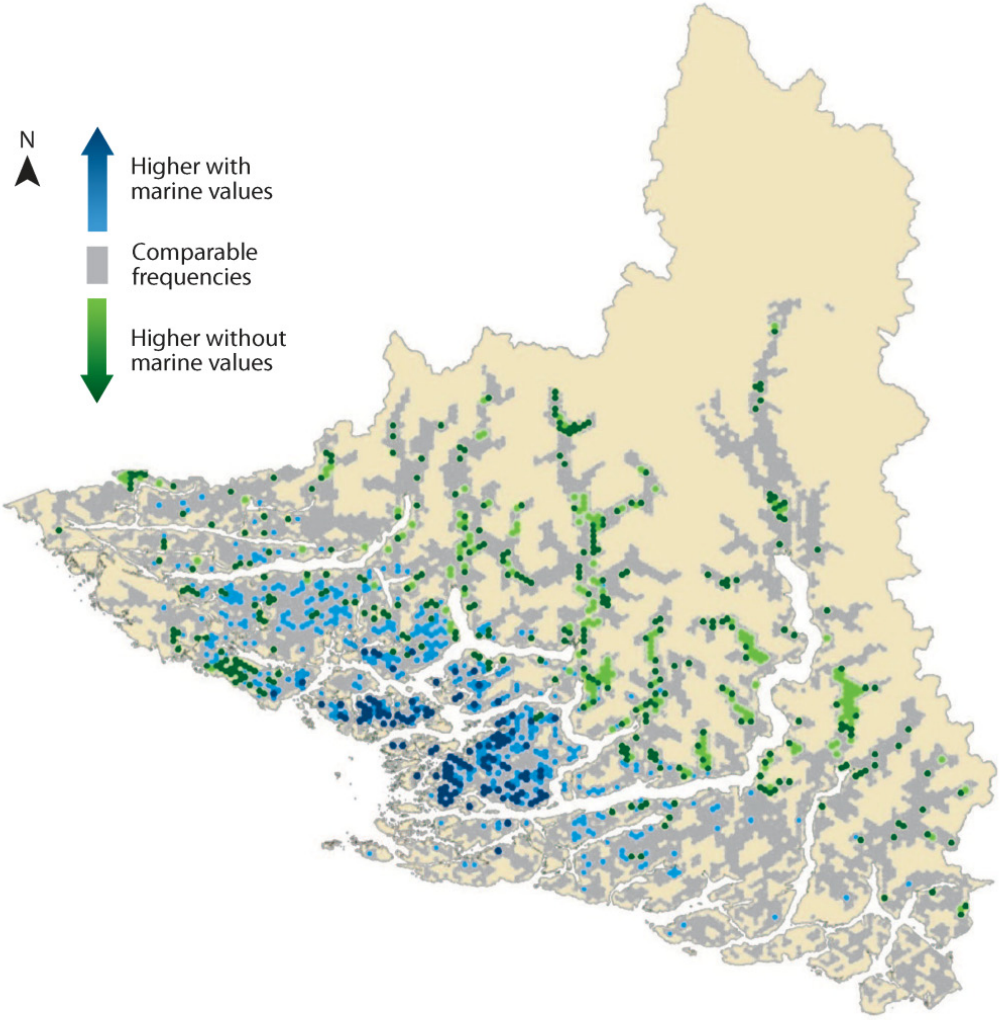

Figure 2.5. Difference in priority areas for conservation of marbled murrelet terrestrial nesting habitat when marine objectives for this species were incorporated or omitted

Conservation alternatives

Systematic conservation planning involves the development and comparison of alternative configurations of areas and alternative actions applied to those areas. Even when designs are presented for public viewing and comment, usually much flexibility exists to reconfigure them and reallocate actions (Cowling et al. 2003).

Approaches to exploring conservation alternatives are diverse, including decision-support systems with input from experts and managers (Geselbracht et al. 2009, TNC 2005), successive designs open for comment and revision (Fernandes et al. 2005), interactive and participatory processes guided by decision-support systems (Dunton 2009; Pressey et al. 2009), and expert-driven processes supported by geographic information systems (Banks et al. 1999, Enriquez-Andrade et al. 2005). These approaches can consider land-sea linkages (e.g., Banks et al. 1999, Kramer & Kramer 2002, TNC 2004). Regardless of the approach, developing conservation alternatives usually involves iterative cycles of analysis in which conservation objectives, design criteria, and spatial data and models are modified or adjusted to suit stakeholdersô preferences and to explore potential solutions under different environmental, socioeconomic, and protection scenarios (Pressey et al. 2009).

Conservation scenarios can reflect variations in objectives (e.g., higher/lower representation), different models of threats (e.g., current/projected, cross-system/single-realm, rapid/moderate change), types of protection and management (e.g., zoning alternatives), and costs, among other factors. Different types and levels of protection and use (e.g., reservation, water management, restoration, land management, fishery regulation) can be linked to specific conservation areas across realms (Banks et al. 1999; Dunton 2009; MPO 2003; TNC 2003). Addressing specific threats, such as high sediment fluxes into coral reefs, might require specific erosion mitigation practices, including reforestation and regulation of logging in catchments (Atherton et al. 2005). Ideally, these strategies should be translated into integrated terrestrial, freshwater, and marine multiple-use conservation systems in which different activities are permitted, prohibited, or regulated consistent with ecological and socioeconomic requirements and constraints (Dutton et al. 1994; Fernandes et al. 2005; Vander Schaaf et al. 2006).

Synthesis

The transition from design to implementation involves aspects of particular relevance to land-sea planning, including: deciding on the conservation areas that will be protected/managed first, involving tradeoffs between land and sea; adapting and communicating planning outputs to stakeholders associated with different realms and across land-sea jurisdictions; and monitoring the achievement of objectives as conservation actions are implemented.

Schedule areas and actions

Conservation plans are seldom fully implemented over short periods, so scheduling of incremental conservation actions is usually necessary to minimise the extent to which objectives are compromised by ongoing attrition of biodiversity (Pressey et al. 2007). Hence, one of the particular challenges of land-sea planning is the requirement for simultaneous scheduling in terrestrial and marine regions so that threats within and between realms can be addressed comprehensively. Klein et al. (2010) described a protocol to guide planners in scheduling conservation actions simultaneously across realms in the coral triangle. The study demonstrated that in any one period, integrated scheduling might require choices between marine areas, between terrestrial areas, or between areas on either side of the shoreline. Deciding between two critical catchments that are equally vulnerable and linked to priority marine areas might require also assessing local values (e.g., terrestrial endangered and rare species and communities). In these cases, experts can play a critical role in determining which areas require immediate attention (e.g., Banks et al. 1999; MPO 2003; TNC 2002), especially when data on the dynamics of local and cross-system threats are limited.

Mainstream technical outputs

Mainstreaming is a critical, but often overlooked, element in conservation planning (Pierce et al. 2005). In land-sea planning, planners should engage with stakeholders and decision makers with influence on or jurisdiction in the land-sea continuum (e.g., Day et al. 2003; Dunton 2009; MPO 2003), in particular those responsible for land-sea issues,

such as catchment-based organisations that consider marine water quality. Mainstreaming for multiple and diverse stakeholders, such as municipal planners, coastal management authorities, and the agricultural and fishing industries, can be complex and requires substantial technical and financial capacity.

Apply conservation actions

Application of actions involves the critical transition from design of conservation areas to implementation of feasible and effective actions on the ground and in the water. Application of actions in land-sea planning will likely require several strategies to address the threats in the land-sea continuum that are implemented via negotiations and arrangements with diverse stakeholders (Baker et al. 2011; Green et al. 2009). Application also requires coordination of administrative arrangements between governmental institutions at different levels (Dutton et al. 1994). Ideally, planning should also deliver a strategy that integrates conservation actions between realms and promotes collaboration of stakeholders from different realms. Existing bodies operating beyond the land-sea boundaries, such as regional catchment management organisations, can significantly contribute to coordination of actions across realms. Inevitably, new data and unexpected constraints and opportunities will be encountered as designs are interpreted and applied. This will likely require extensive adaptation of designs to accommodate these changes (**Figure 2.3**).

Monitor achievement of objectives

Monitoring is a critical component of an adaptive planning process (Baker et al. 2011). It informs planners about the effectiveness of applied actions in achieving objectives across realms, so that actions, and the areas where they are to be applied, can be adjusted accordingly (Dunton 2009; Dutton et al. 1994; Fernandes et al. 2005). Even though they are a critical component of effective planning, monitoring activities either have not been considered or have been developed for a single realm or separate realms. Rarely has monitoring included species and habitats in different realms as well as sources and impacts of cross-system threats. Notable exceptions are the system-wide monitoring program linked to the Mission-Aransas land-sea planning initiative (http://cdmo.baruch.sc.edu: Dunton 2009) and the water quality strategy for the great barrier reef marine park (Carroll et al. 2012).

Challenges for the future

Scientists, managers, and conservation practitioners have called for an integrated approach to managing terrestrial, freshwater, and marine socioecological systems (Dutton et al. 1994; Gordon 2007; Stoms et al. 2005). Different approaches to land-sea integration have been followed (for a description of integration levels, see Tallis et al. 2008), including concurrent area prioritisation exercises for terrestrial and marine systems that are later assembled to build a single plan (e.g., Floberg et al. 2004); simultaneous planning, in which land and marine features are targeted at the same time and some measure of conservation cost is minimised for both realms simultaneously (e.g., TNC 2005); and explicit incorporation of cross-system threats (e.g., Tallis et al. 2008) and/or land-sea processes (e.g., Lombard et al. 2007) in the prioritisation process. Within these approaches, different elements of the framework described above have been incorporated, but no planning exercise has addressed all.

Expert opinion (based on a survey I conducted) indicates that few components of the operational framework I outlined have been incorporated into land-sea planning (**Figure 2.6**). In addition to the particular needs for improvement illustrated in **Figure 2.6** and previously discussed, I summarise below three key challenges to land-sea planning based on reviewed studies, expert opinion, and my own experience.

Figure 6. Expert opinion on the level of land-sea integration for each of the components of the operational framework in **Figure 2.3**. Integration within components of the first planning phase was generally rated between low and medium. In the second planning phase, two components were considered to have achieved relatively good integration: planning for cross-system threats (a common purpose of land-sea planning) and the use of decision-support tools. Also within the second phase, incorporation of climate change and explicit costs seem to deserve special attention. Experts generally identified components in the third phase as involving poor integration between land and sea. These included scheduling, which had the lowest rating of all components in the planning process. The horizontal black bars represent the average value of integration assigned by experts ($N = 36$) to each component (1 = low, 2 = medium, and 3 = high integration); error bars denote the 95% confidence interval for the mean ($\alpha = 0.05$, $p < 0.0001$).

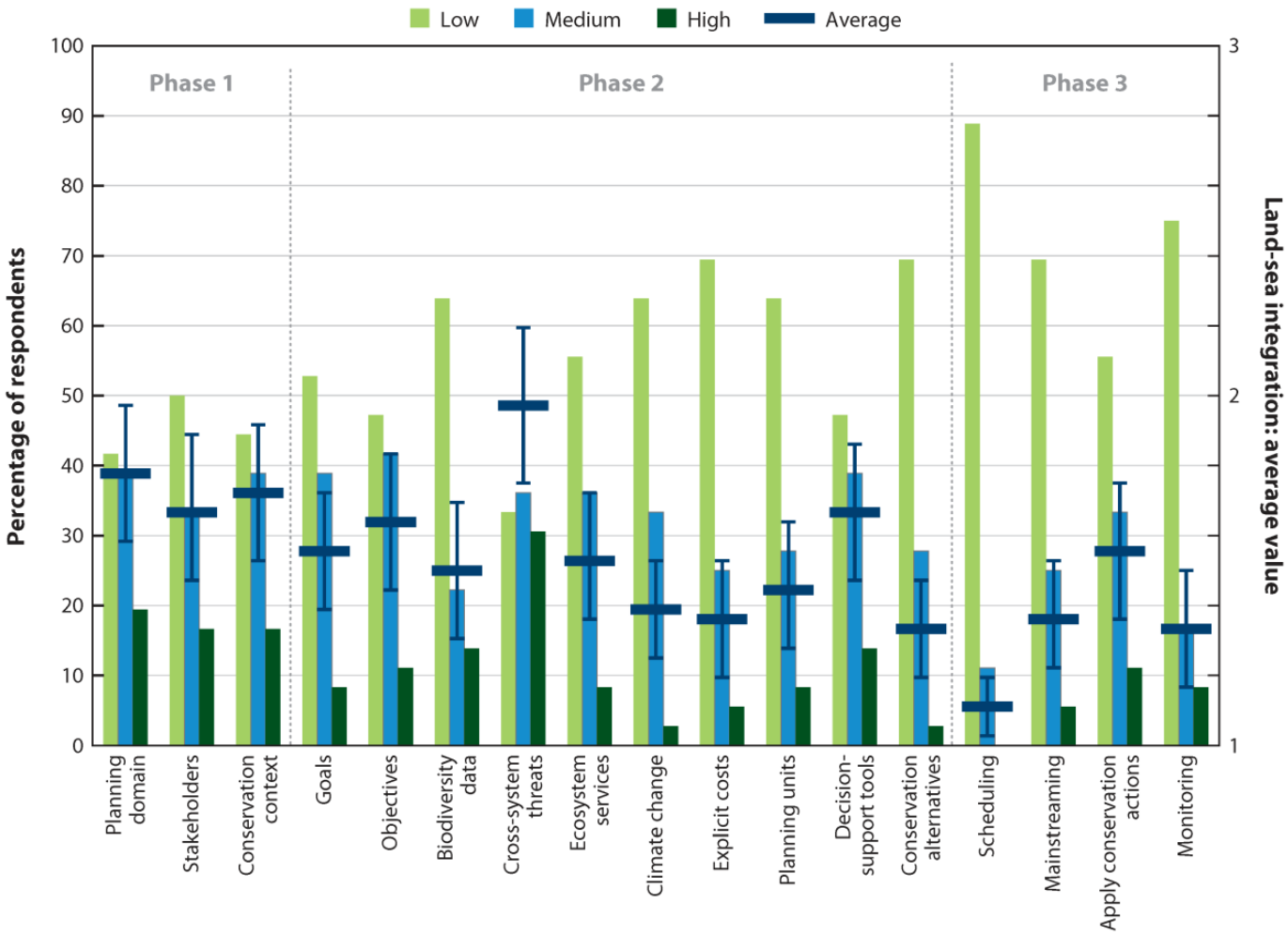

Figure 2.6. Expert opinion on the level of land-sea integration for each of the components of the operational framework

Partial integration of cross-system threats

Although cross-system threats are widely cited as a major concern for marine conservation and often considered in land-sea planning exercises, these have been only partially incorporated into prioritisation processes. The impacts of threats with marine origins on upstream ecosystems are poorly studied and not yet incorporated into planning (Green et al. 2009). Data required to model threats originating on land (e.g., pollutant loads and zones of offshore influence) are usually incomplete, available only for short periods, coarse in resolution (Atherton et al. 2005; Kramer & Kramer 2002), or inadequate for planning purposes (TNC 2002). Data limitations can be overcome through the use of simplified models to assess exposure to cross-system threats (e.g., Maughan & Brodie 2009) or expert-based delineation of high-risk areas (Banks et al. 1999). However, simplified data limit the development of adequate conservation actions to mitigate these threats, and they could result in erroneous conclusions and poor management recommendations. More advanced methods of catchment and ocean circulation modelling (e.g., Paris & Cherubin 2008) can be employed for land-sea planning if resources, data, and expertise are available. In most cases, however, lack of spatial data to calibrate and validate the models remains a major challenge. Developing and integrating models into a decision-support system can be time- and labour-intensive (Bohnet et al. 2011; Crist et al. 2009; Dunton 2009;), and trade-offs between quality of results and investment in data and modelling seldom have been assessed .

Competing objectives

Integrated land-sea planning must address many potentially conflicting objectives. Some studies have found coincidence in the spatial distribution of priorities arising from different objectives, but others have not. Differences in the spatial distribution of conservation priorities for terrestrial biodiversity and ecosystem services (Chan et al. 2006) and of freshwater and terrestrial priorities (Amis et al. 2009) suggest that values within coastal catchments derived from terrestrial conservation objectives (e.g., connectivity between vegetation fragments) will not necessarily be spatially correlated with those for marine conservation (e.g., reducing pollutant loads at river mouths). However, an integrated study from Florida showed how objectives for both coastal hazard mitigation and biodiversity conservation can be jointly met with little change in

the extent of priority areas and with increased spatial efficiency compared with addressing the two sets of objectives separately (TNC 2007). When choices are necessary between objectives, decision makers face complex problems. Some methods have been developed to assess the necessary trade-offs (Klein et al. 2010; Moffett & Sarkar 2006), but more tools are needed, especially to work interactively with stakeholders.

Institutional structures

A fundamental barrier to integrated land-sea planning is the lack of coordination of institutions that support natural resource management. Typically, different institutions, departments, or groups govern marine and terrestrial natural resources (Cicin-Sain & Belfiore 2005). This is also true within a single realm. For example, decisions about fisheries management, marine ecosystem conservation, and mineral extraction have often been made independently by different sectors within a government (Crowder et al. 2006). This segregation is apparent across, and even within, agencies, NGOs, and scientific institutions. In a given region, conservation NGOs often have marine and terrestrial planning teams that work with only limited exchange of information. A similar segregation is common in the scientific community, as different research groups, conferences, journals, and grant schemes exist for marine and terrestrial scientists. The divided nature of natural resource management affects policies, budgets, and goals. Effective land-sea planning will require substantial changes in institutional and organisational structures to allow for more cohesive and integrated planning to occur.

Conclusion

Incorporating land-sea linkages in conservation planning presents big challenges but also unique opportunities, especially given the many marine spatial planning initiatives currently underway (e.g., under the Convention On Biological Diversity, United Nations Educational, Scientific and Cultural Organisation, World Ocean Council, U.S. National Ocean Council). Gaps in data and knowledge are open niches for theoretical and applied research in multiple disciplines, including ecology, informatics, economics, politics, and sociology. Planning for land-sea processes requires the development of

approaches and tools to engage with multiple stakeholders and incorporate their views and values. Conservation practitioners will be required to formulate practical approaches to make best use of available information. Learning by doing and communicating lessons learned from each attempt will advance integrated land-sea planning. Integration of information on different realms can help to bring together scientists, managers, and practitioners from diverse disciplines and sectors who have previously focused on different realms. Bridging interdisciplinary and inter-sectoral gaps is critical to generating a coordinated response to multiple, interactive, and cumulative threats to biodiversity, which in turn can lead to stronger and more effective conservation initiatives. Making these links will require substantial institutional and policy changes in the management of natural resources. Interdisciplinary and cross-realm collaboration should therefore continue to be in the core of the programs of international agreements such as the convention on biological diversity. Therefore, further development of the methodological foundations for integrated land-sea conservation and effective communication of these findings to decision makers are important contributions for conservation planners and scientists to make.

Chapter 3. Marine conservation planning in practice: lessons learned from the Gulf of California

Abstract

Overfishing, pollution, coastal development and climate change threaten marine biodiversity globally and compromise the services that marine ecosystems provide. Systematic conservation planning (SCP) provides a framework to identify areas where actions can be effective in addressing these threats, while minimizing the costs of interventions. I investigated the application of SCP in the Gulf of California, a marine biodiversity hotspot where seven spatial prioritization exercises have been undertaken. The review of planning exercises showed that the use of SCP methods has progressed slowly (gaps include planning for land-sea connections and ecosystem services) and highlighted benefits and difficulties of applying SCP principles and tools. Despite some convergence, I found important spatial differences in priorities between plans. Convergence was evident in well-studied shallow and benthic marine ecosystems. There were also important differences related to the planning approach, methods and extent. Divergence between methodological and spatial similarities between plans supports the idea that various elements, in addition to data and objectives, play an important role in defining the distribution of conservation priorities. According to expert opinion, the implementation of new marine protected areas (MPAs) in the region has been influenced by some of the planning exercises. However, uptake of planning outputs has progressed slowly due to multiple factors (e.g., conflicting mandates and interests between organizations, limited technical capacities and resources, insufficient political commitment). Apart from MPAs, other benefits of planning included: developing institutional skills and knowledge; improving collaboration and coordination between organizations (including agencies, and local, regional and national NGOs); converging on the need to assess priorities for marine conservation in regional context; and building trust among organizations. The existence of multiple marine conservation plans in the Gulf of California also highlighted some of the complexities and benefits of having multiple sets of priorities.

Introduction

Biodiversity conservation of marine ecosystems remains a global priority (Spalding et al. 2010) because fish stocks continue to decline (Worm et al. 2009), eutrophication of coastal areas is ongoing (Howarth 2008), and extinctions of marine species continue (Sala & Knowlton 2006). The services provided by marine ecosystems (e.g., food provisioning, water quality, coastal protection) are also being compromised (Aburto-Oropeza et al. 2008; Worm et al. 2006). Ongoing conservation efforts, including marine protected areas (MPAs) (Lester et al. 2009; McCook et al. 2010; Spalding et al. 2010), fisheries management (Mora et al. 2009; Worm et al. 2009), and ecosystem-based spatial planning and management (Foley et al. 2010; Leslie & McLeod 2007), are providing some benefits to protect marine biodiversity, but are not yet sufficient to halt or reverse the declines. The expansion of threats to the oceans and the limited resources available for conservation thus call for effective, transparent, and cost-effective strategies to protect biodiversity and maximise benefits for human livelihoods.

Systematic conservation planning (SCP) provides a framework to identify areas where conservation actions can be effective at achieving conservation objectives, while minimizing the costs of interventions (Pressey & Bottrill 2009). Benefits beyond guiding effective investment in conservation areas include increasing socio-ecological knowledge relevant to biodiversity conservation, building institutional and individual capacity, facilitating collaboration among stakeholders, and increasing awareness of the need for urgent action to prevent biodiversity loss (Bottrill et al. In review; Pressey & Bottrill 2009). Conditions for successful implementation of actions in conservation areas ï as identified through SCP exercises ï have been shown to include integration of ecological and socioeconomic criteria into designs of conservation areas (Fernandes et al. 2005; Green et al. 2009), stakeholder ownership and support (Gleason et al. 2010), input from managers and experts (Fernandes et al. 2009), effective participation and engagement with local communities (Baker et al. 2011; Game et al. 2010), consideration of practical aspects of implementation (e.g., simple boundaries of conservation areas) (Fernandes et al. 2005; Lombard et al. 2007), adaptive management (McCook et al. 2010), transparency during planning, and effective communication strategies (Day 2002).

Given the increasing recognition of the value of systematic planning for marine conservation, research in the field is advancing rapidly (Leslie 2005). The initial limited focus on representation of biodiversity patterns is expanding to include planning for ecological processes and socioeconomic considerations. Major recent advances include planning for several kinds of dynamics and interactions: the persistence of dynamic oceanographic processes and pelagic biodiversity (Game et al. 2009; Grantham et al. 2011; Lombard et al. 2007), connectivity between different habitats (Beger et al. 2010b; Edwards et al. 2010), adaptation to climate change (Green et al. 2009; McLeod et al. 2009), land-sea integration (Klein et al. 2010; Tallis et al. 2008), and socioeconomic considerations (Baker et al. 2011; Ban & Klein 2009). These advances have guided some recent conservation strategies (Baker et al. 2011; Green et al. 2009).

While theory in conservation planning is developing quickly, there has been no assessment of the influence of new ideas on applications of marine conservation. Previous studies have reviewed aspects of MPA design that led to successful conservation actions (Lundquist & Granek 2005; Osmond et al. 2010) and there are retrospectives on particular planning exercises, e.g., the Great Barrier Reef Marine Park rezoning (Fernandes et al. 2005) and the Northern Central California MPA network design (Gleason et al. 2010). However, I am aware of no previous studies that assessed multiple marine planning initiatives within a single region, using a comprehensive SCP framework for review and comparison.

The Gulf of California is ideal for comparative studies of marine conservation planning, with seven marine planning exercises undertaken in the past 15 years. This study investigated how SCP principles have guided marine prioritization in the region, identified missing components of the planning process, and discussed the benefits and difficulties of applying SCP principles and tools. The similarities and differences between planning exercises were examined in terms of data, methods and outputs, how identified priorities match the existing MPA system, and whether plans have guided conservation and management actions. Difficulties and opportunities for conservation planners and practitioners are discussed when multiple planning initiatives coexist.

Methods

Study region

The Gulf of California is a marine conservation hotspot (Olson & Dinerstein 2002; Roberts et al. 2002), globally recognised for its outstanding marine biodiversity (Brusca et al. 2005; Lluch-Cota et al. 2007). Oceanographic processes, including gyres, fronts, and upwellings (Lav²n & Marinone 2003), contribute to high levels of primary production that sustain rich coastal and marine communities and important fisheries (Aburto-Oropeza et al. 2008; Beman et al. 2005; Lluch-Cota et al. 2007). The Gulf is an important feeding, breeding and nursery area for 36 marine mammals (Lluch-Cota et al. 2007; Morgan et al. 2005), including the critically endangered and endemic vaquita, *Phocoena sinus* (Jaramillo-Legorreta et al. 2007). The Gulf supports large portions of the worldôs populations of several marine bird species (Lluch-Cota et al. 2007) and provides critical habitat for migrating birds (Glenn et al. 2006). The islands of the Gulf of California have ~655 plant species, 28 of which can be found nowhere else (Rebman et al. 2002). Comparable levels of endemicity have been recorded for marine fauna, including 766 invertebrate and 87 fish species (Brusca et al. 2005).

The Gulf of California retains most of its natural values, but rapid expansion and intensification of human activities are threatening its biodiversity and associated ecosystem services. Human population density is relatively low in the catchments draining into the Gulf of California, but rapidly increasing, with associated increases in threats to the marine environment (Lluch-Cota et al. 2007). While the western coast remains comparatively undisturbed, many eastern coastal areas and catchments draining into the Gulf are affected by aquaculture, industry, agriculture, and urban development (Beman et al. 2005; Lluch-Cota et al. 2007; P§ez-Osuna & Ruiz-Fern§ndez 2005). Intensive use of pesticides in agricultural areas is also a concern (Galindo-Reyes et al. 1999; Gutierrez-Galindo et al. 1992; Ni¶o-Torres et al. 2010). Alteration of freshwater and sediment inputs from the Colorado River has modified the habitat for commercial and endangered species (All 2006; Carriquiry & S§nchez 1999; Rowell et al. 2008). Saltgrass marshes and mangrove forests are being reduced and degraded by coastal development, land-based pollution, and shrimp farms (Aburto-Oropeza et al. 2008; Glenn et al. 2006; INE 2005). Deposition of atmospheric pollution and point sources of

mercury (e.g., mining, urban and industrial sewage) are also affecting marine food webs (Garc²a-Hern§ndez et al. 2007). Finally, the dramatic increase in fishing effort and inadequate management of fisheries have led to substantial declines of multiple fish stocks, changes in populations of targeted species, and composition of fish communities (Sala et al. 2004), which in turn have affected regional and local economies (Erisman et al. 2010).

Data collection and sources

Spatial marine conservation planning exercises were the focus of this study, excluding policy, regulation, and education initiatives. Seven planning exercises were identified and reviewed that prioritised coastal or marine conservation areas in the Gulf of California. The extents of these exercises, developed between 1998 and 2006, ranged from sub-continental (B2B: Morgan et al. 2005), to national (RMP: Arriaga-Cabrera et al. 1998; SPM: CONABIO 2007a), regional (i.e., Gulf-wide) (CSGC: Enr²quez-Andrade et al. 2005; ERA: Ulloa et al. 2006), and sub-regional initiatives (i.e., covering portions of the Gulf) (BCP: Enr²quez-Andrade & Danemann 1998; MRN: Sala et al. 2002). **Table 1** contains full names for acronyms and brief descriptions of the exercises. Information sources included published and unpublished reports and digital information. A digital map of marine priority conservation areas (hereafter ñpriority areasò) and details of datasets were not available for one plan (BCP). This plan included only the Baja Peninsula and a portion of the Gulf of California, and defined priority areas based on modifications to the National Marine Priority Regions (RMP). For these reasons, spatial analyses were focused on the other six planning exercises. However, key elements of this seventh planning process were summarised and discussed. A comprehensive description of the methods and planning stages for each plan, as well the maps of priority areas, are provided as **Supplementary Material**.

Review of approaches to marine conservation planning

Key aspects of a systematic conservation planning framework developed by Ćlvarez-Romero et al. (2011) were used to identify commonalities and differences in approaches and methods between the seven marine conservation plans (**Table 1**), and to identify missing elements or stages (see **Supplementary Material** for details). This framework

includes key components in planning (e.g., conservation goals and objectives, stakeholder participation, threats, and costs) and emerging issues in SCP (e.g., planning for ecological processes, ecosystem services, and land-sea interactions).

Assessment of methodological and spatial similarity

Methodological similarity between planning exercises was calculated based on the Jaccard coefficient (which adds up the total number of items that are different, excluding instances of zero ties), focusing on conservation objectives and data because these were assumed to be most influential in the selection of priority areas. First, conservation objectives and data for features of interest were coded as binary attributes (1 = used; 0 = not used). When planning exercises used datasets differently (e.g., extent, resolution, date) or when data were grouped or disaggregated differently, data types were matched with the most equivalent elements in other exercises or grouped in the same way as in other exercises. Based on an overlay of the maps (a.k.a. portfolios) of priority areas (hereafter ñpriority mapsò), areas of coincidence were identified and spatial similarity between pairs of priority maps were measured with the Kappa statistic using the Map Comparison Kit (Visser & de Nijs 2006). Kappa similarity is adequate and commonly used for assessing agreement between categorical maps (Edwards et al. 2010; Wilson et al. 2005b). The Jaccard coefficient and Kappa statistic were displayed with similarity phenograms based on hierarchical clustering, also applied previously (Ban 2009; Weeks et al. 2010b). In addition, the spatial configuration of priority maps (i.e., number, size, perimeter, and spacing) and the sizes of the planning domains were also compared. A second spatial analysis focused on how areas of convergence between the six planning exercises were related spatially to existing MPAs (CONANP 2010), marine ecoregions (CEC 2006), and selected ecosystems and seascape features (CONABIO 2008; Lugo-Hubp & Fernandez C· rdova 1990; Ulloa et al. 2006; UNEP-WCMC 2010).

Evaluation of achievement of goals and benefits of planning

Interviews were carried out with 13 experts, selected based on their expertise in marine conservation in the region and their involvement in one or more of the planning processes discussed here. During these interviews, experts identified both reported and

unreported planning goals and indicated the extent to which these goals had been achieved. Further, the extent to which planning outputs guided or influenced spatial implementation of conservation actions was assessed, including which actions were more common. Experts also identified the main factors perceived to facilitate or constrain actions. In addition, other perceived benefits of the planning processes (besides spatial actions) were identified, as well as the existence of any formal assessments of outcomes, and intentions for revising plans. Interviews were supported by a written survey (**Supplementary Material**).

Results

Approaches to marine conservation planning

All seven planning exercises included fundamental elements of systematic conservation planning (e.g., involving stakeholders, setting conservation objectives, assessing threats and conservation costs, and delineating new priority areas), but differed widely in the way they addressed these elements (see **Table 1** and **Supplementary Material**). Notable differences included the way costs were incorporated (e.g., least-cost solutions vs. prioritization of highly threatened areas) and methods for identifying priority areas (e.g., expert-based vs. software). In the following sections the most relevant similarities and differences are summarised and discussed.

Stakeholders

Stakeholder participation varied considerably between planning exercises (**Table 1**). Participation of researchers was prominent in all cases, and input from agencies, conservation non-governmental organizations (NGOs) and MPA managers was substantial in most exercises. Two planning processes (B2B and SPM) also included some representatives of the fishing and tourism sectors. A contribution by resource users was to provide information on areas of economic importance, such as fishing grounds (e.g., Ulloa et al. 2006). Stakeholder involvement in one planning exercise (MRN) was restricted to academics and conservation NGOs. In the four expert-based

processes, representatives from agencies, academia and NGOs actively participated and contributed to different stages of the planning process.

Conservation objectives

Approaches to setting conservation objectives were diverse (**Table 1**). A coarse/fine-filter approach to identifying conservation features (i.e., targeting communities/ecosystems and individual species/features independently) (Groves et al. 2002) was taken in most exercises, but there were also important differences. Quantitative objectives were set for only two planning exercises (MRN and ERA), both of which also defined separate objectives for conservation features occurring in more than one bioregion. Qualitative objectives were common to all planning exercises. Qualitative objectives included selecting areas with high species richness and abundance or endemic, endangered or protected species, and maintaining connectivity between areas.

Threats and costs

There were major differences between planning exercises in the treatment of threats and costs. In two exercises (SPM and CSGC), important threats to coastal and marine ecosystems were identified independently of the prioritization process and therefore had no influence on the selection of priority areas. One exercise (CSGC) mapped current and near-future (5-year projection) threats to inform the feasibility of implementing conservation actions, as well as to identify potential areas of conflict. Threats and potential mitigation strategies were documented for the national marine priority regions (RMP) after these were delineated by experts. For the B2B exercise, a qualitative ranking of the intensity and trend of threats was developed by experts to identify areas with both threats and conservation opportunities (e.g., ongoing conservation or management efforts). In contrast, the two planning exercises supported by conservation planning software (MRN and ERA) used data on threats to achieve objectives while minimizing conflicts and/or conservation costs (Ban & Klein 2009) by placing priority areas away from highly threatened or heavily used (e.g., fishing) areas, where possible.

Emerging issues

Uptake of recent advances in SCP (e.g., planning for ecosystem services, land-sea interactions, and climate change) was limited, with the exception of ecological processes (see **Supplementary Material**). Generic design criteria such as minimum size and connectivity guided the selection of priority areas to address some processes. To maintain larval dispersal, the MRN exercise used software to ensure that distances between priority areas did not exceed 100 km. For the B2B exercise, experts applied generic design criteria, supported by available information on processes such as migration routes and areas of high productivity, to delineate priority areas. Ecological processes linking land and sea, such as nutrient flows from rivers, were considered by experts in the national marine gap analysis (SPM), but no spatial data were available. Cross-system threats (e.g., land-based pollution and coastal development) were considered by experts in B2B and, in the ERA exercise, simple spatial models of land-based threats (i.e., multiple-ring buffers associated with higher conservation costs) were incorporated. Only one planning exercise (CSGC) explicitly targeted ecological linkages between terrestrial, freshwater and marine priority areas. In the national marine gap analysis, a *post hoc* integration with terrestrial and freshwater priorities was recommended for the future. None of the studies scheduled priority areas to sequence investments for conservation actions through time, but one exercise (SPM) assigned priority levels, based on expert advice, to priority areas after they were delineated and another (ERA) recommended scheduling.

Similarity and spatial coincidence between planning exercises

Methodological similarity

Clustering of exercises based on objectives and datasets (**Figure 1a**) showed that the exercise emphasizing protection of fish (MRN) was very different to the others, which also considered other taxonomic groups and oceanographic processes. Likewise, similarity values between the sub-continental exercise (B2B) and other plans were relatively low. B2B focused on species of conservation concern throughout North America, not on those more typical of the Gulf, and also considered processes operating over larger geographic extents (e.g., migratory corridors) than the Gulf. Common to

both MRN and B2B was their focus on a few species and key habitats. Also notable was the similarity between the two regional-scale exercises (CSGC and ERA) which, despite differences in methodology (expert-based vs. software-driven), shared many conservation objectives, and used similar data. Similarity between the two national-scale exercises (RMP and SPM) was relatively low. My analysis placed SPM closer to the regional plans and RMP somewhere between the regional and the sub-continental exercises. This is consistent with both B2B and RMP identifying regions, rather than specific areas, for marine conservation and their use of data appropriate for that purpose.

Spatial similarity

Overall spatial similarity (**Figure 3.1b**), accounting for spatial overlap of priority areas and their relative proportions in the region, revealed two subgroups (CSGC-B2B and ERA-SPM) and two planning exercises that were substantially different from the rest (RMP and MRN). Spatial similarity based only on the locations of priority and non-priority areas (K-loc), and not their relative proportions (**Figure 3.1c**), showed higher values but also a different clustering pattern. For instance, MRN ï isolated based on overall Kappa ï formed a new subgroup with CSGC based on K-loc. This cluster could result from priority areas from MRN being almost entirely contained within CSGC and, in both exercises, the preferential location of priority areas in coastal and nearshore environments. On the other hand, the subgroup formed by ERA and SPM based on overall spatial similarity (**Figure 3.1b**) remained in the location-only phenogram (**Figure 3.1c**), but was stronger for location-only, possibly due to the inclusion (with minor adjustments) of some of the ERA priority areas into the SPM priority map.

Figure 3.1. Similarity phenograms for marine planning exercises constructed through agglomerative hierarchical clustering analysis (unweighted pair-group average). B2B: Baja-to-Bering; RMP: Marine Priority Regions; SPM: Marine Priority Sites; CSGC: Coalition's Important Conservation Areas; ERA: Eco-regional Assessment; MRN: Marine Reserves Network. (a) Methodological similarity based on planning objectives and datasets (Jaccard coefficient); (b) Overall spatial similarity based on location of priority areas and relative proportions (Kappa statistic); and (c) Location-only similarity (K-loc statistic: Visser 2004). Spatial similarity varied, but general strength of

agreement between pairs of priority maps was fair based on overall similarity (Mean Kappa = 0.24) and moderate for similarity in location-only (Mean K-loc = 0.52). A value of 1 means identical priority maps, while 0 indicates no more overlap than expected by chance. A significance analysis based on random constraint match models (Visser & de Nijs 2006) indicated that Kappa values between pairs of maps were significant when compared to similarity with random maps (0.066-0.074), $p<0.001$.

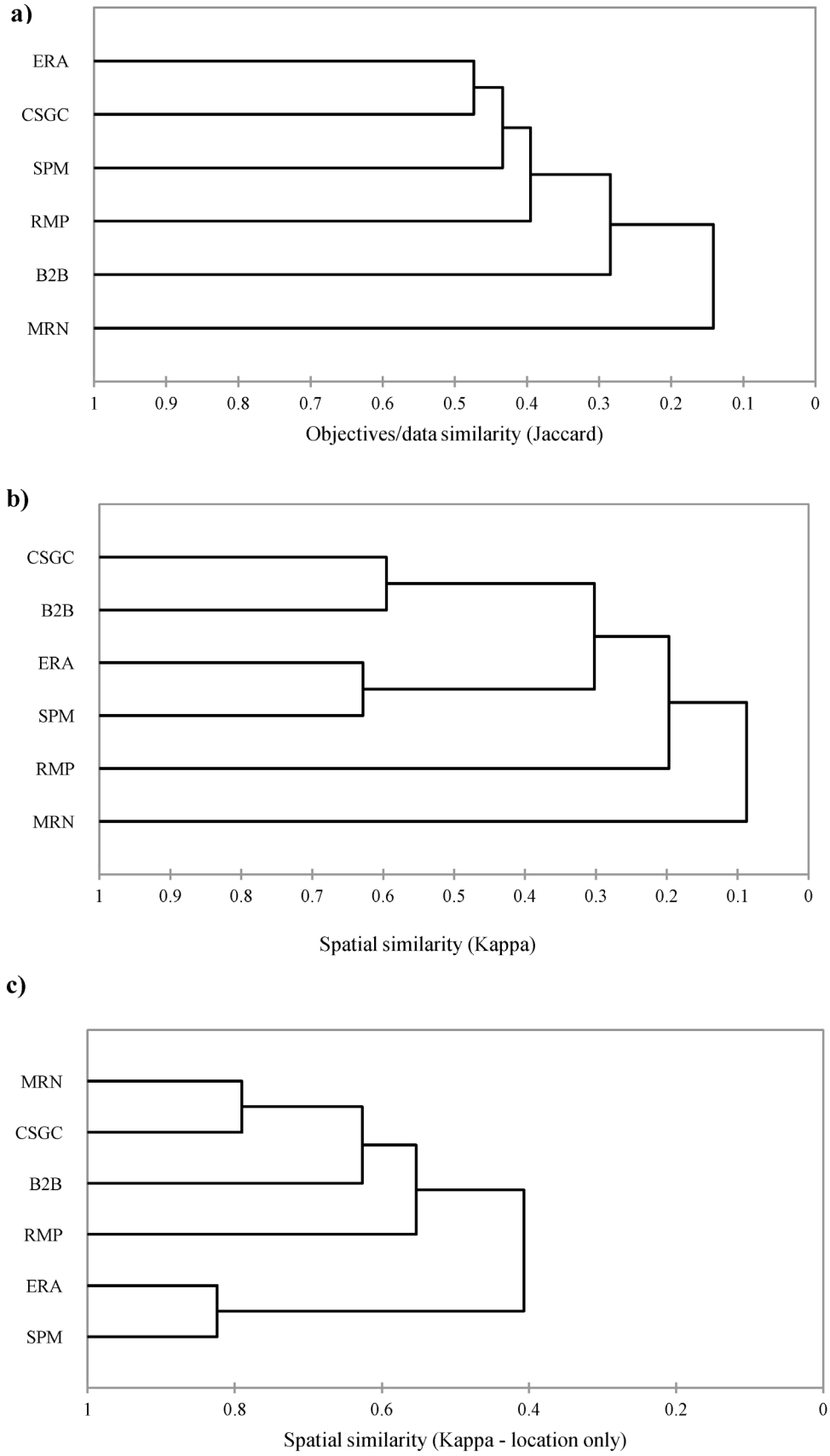

Figure 3.1. Similarity phenograms for marine planning exercises constructed through agglomerative hierarchical clustering analysis

Configuration of priority areas

There was an asymptotic increase in the mean size and perimeter of priority areas as the extents of planning domains increased, with significant variation within each plan (**Figure 3.2a, b**). There was an inverse relationship between the number of priority areas and the extent of the planning domain, though the strength of this relationship was fairly low (**Figure 3.2c**). Excluding the values corresponding to MRN ï the exercise that focused on fish protection (formed by relatively few, small priority areas intended to form a network of no-take zones) ï appreciably increased the fit between number of priority areas and extent of domain. The tendency for larger planning domains to be associated with larger and fewer priority areas (**Figure 3.2a-c)** was related to coarser data resolution, larger planning units being assessed and compared for conservation, and more extensive targeting of ecological processes. The extent of the planning domain was positively correlated with the separation between priority areas and therefore with lower potential connectivity (**Figure 3.2d**). Again, excluding MRN improved curve-fitting because its proposed marine reserves were considerably smaller than all other exercises, and thus were relatively widely spaced relative to the extent of the planning domain. Also evident from **Figure 3.2c, d** is that RMP identified fewer and significantly more separated priority areas than SPM. In these aspects, RMP was more similar to the sub-continental exercise (B2B) than to the other national plan, consistent with the methodological similarity discussed above.

Figure3. 2. Comparison of selected features of priority maps in relation to the full planning domain extent (values on Y axes correspond to priority areas within the Gulf of California only): a) mean size and b) mean perimeter of priority areas; c) total number of priority areas comprising each priority map; and d) median separation between nearest priority areas (use of median responds to highly skewed distributions). Points corresponding to the national-scale exercises are marked by blue (SPM) and red/dashed (RMP) circles. Fitted trend lines correspond to power (a, b), exponential (c) and linear (d) types; light-grey lines show an improved fit when MRN is excluded from the analysis (see text). Error bars represent the 95% C.I. for the mean ($p<0.0001$) estimated by Monte Carlo simulation (a, b) and the 1st and 3rd quartiles (d).

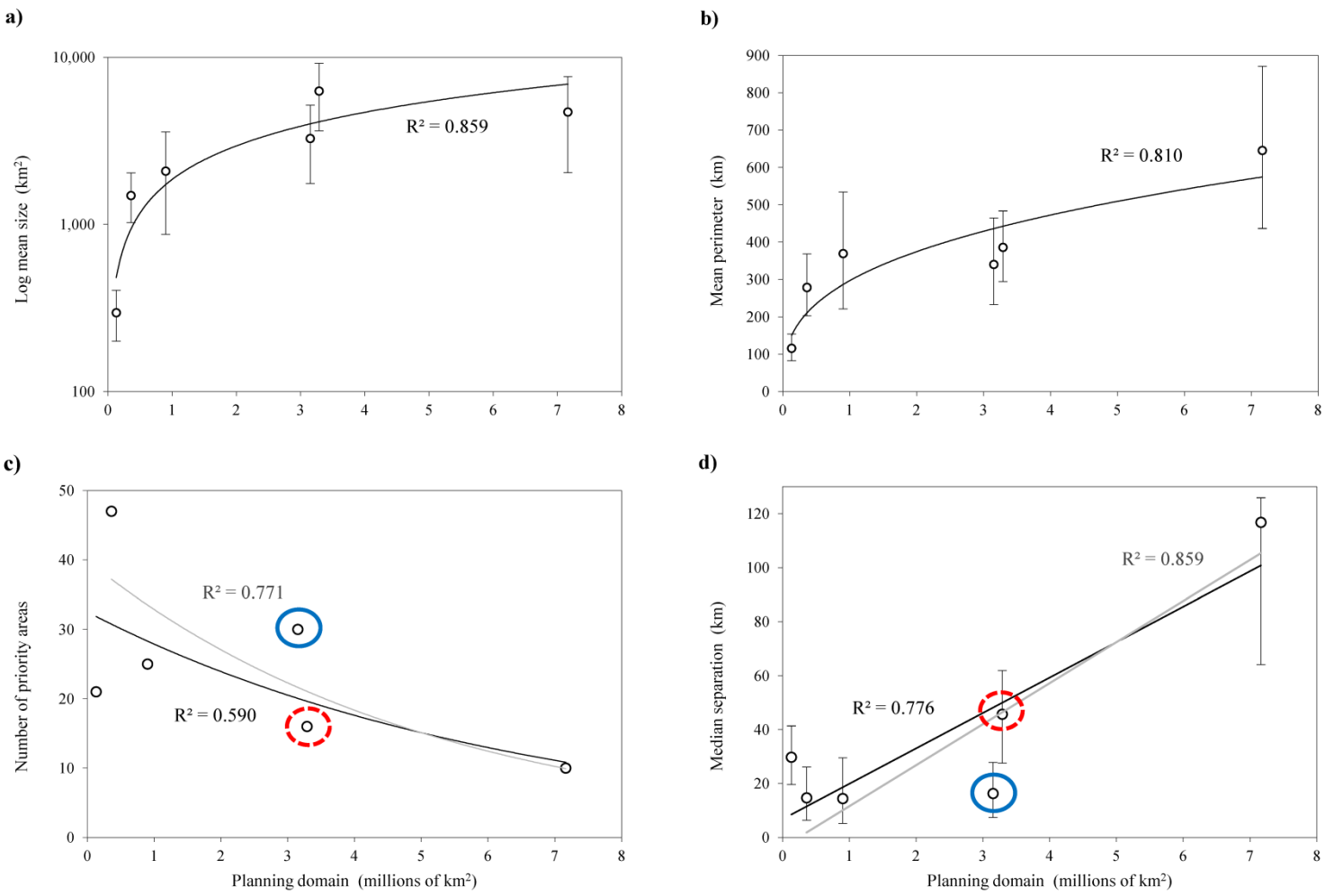

Figure 3.2. Comparison of selected features of priority maps in relation to the full planning domain extent

Overall spatial coincidence, ecological representativeness and MPA coverage

The combined (summed) priority maps covered ~100,000 km^2, but relatively little of the summed area was consistently identified as having priority for marine conservation. About 31% of the summed area was identified by at least 4 plans and only 3% was selected by all plans. This was reflected in the negative relationship between the frequency of spatial coincidence and the total priority area in each frequency (**Figure 3.3**). Conversely, the percentage of existing MPAs covered by priority areas increased with the frequency of spatial coincidence. This means that a large proportion of the areas consistently identified as priorities have also been granted some type of legal protection (**Figures 3.3, 4**), although, in some areas protection preceded prioritization exercises. Spatial coincidence was generally higher in the northern Gulf and western coast (**Figure 3.4**).

Figure 3.3. Frequency of spatial coincidence between the priority maps (number of plans identifying an area as a priority) and two variables: total area covered by each frequency of spatial coincidence (blue dots and line); and proportion of the total extent of existing MPAs identified as priority areas (green squares and line). Fitted lines correspond to 2nd-order polynomial regressions.

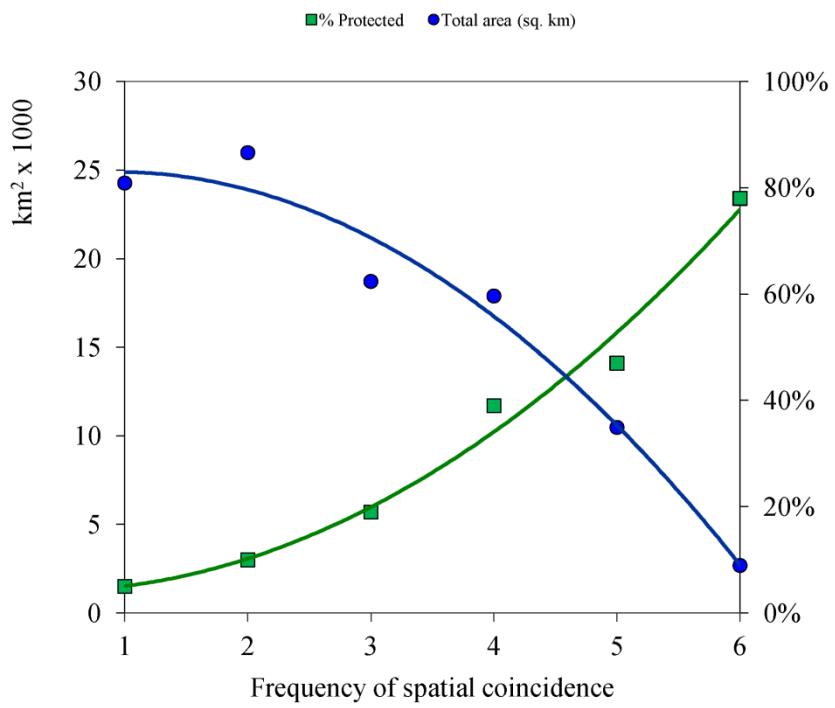

Figure 3.3. Frequency of spatial coincidence between the priority maps

Representation of marine ecoregions varied markedly among plans (**Figure 4**). About one-third of the fourteen ecoregions within the Gulf of California were covered almost entirely by selected priority areas of most plans, including the highly productive Upper Cortezian Inner Neritic and Midriff Islands Straits. In contrast, significant portions of other ecoregions, including the Slope and Basins, the Guaymean Neritic, and the East Pacific Rise were only partially incorporated into priority maps.

Figure 3.4. Spatial coincidence of priority areas between planning exercises in the Gulf of California. For overall frequencies, darker areas were identified as priorities in more plans. Pie charts represent the percentages of each marine ecoregion in three classes of frequency of spatial coincidence across plans, from none (dark blue) to most plans (red).

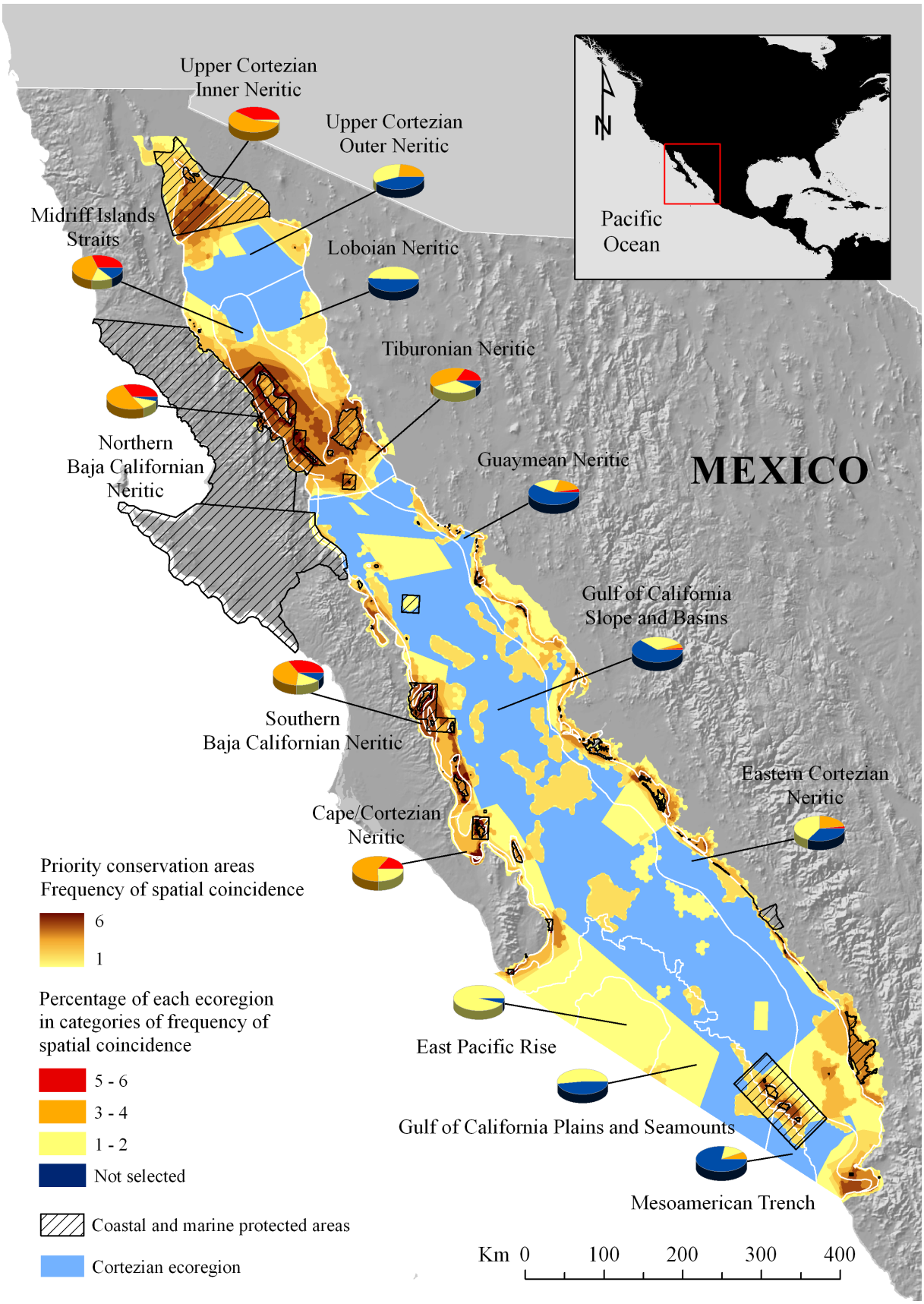

Figure 3.4. Spatial coincidence of priority areas between planning exercises in the Gulf of California

Priority areas were concentrated in coastal and shallow areas (<300m depth), with frequency of prioritization decreasing with depth. This is evident in the biased coverage of geomorphological features (**Figure 3.5a**), with priority areas more often identified on the continental shelf. Features such as trenches, submarine canyons, and valleys were often omitted. Ecosystems were also unevenly represented (**Figure 3.5b**). Coral reefs, seagrasses, and rhodolith and *Sargassum* beds were more often included in plans, while others such as seamounts were rarely included, probably because of limited information. Mangroves were included to some extent in most plans, but the portion of mangrove covered by high coincidence of priority areas (i.e., overlaps > 4 exercises) was minimal. Partial inclusion of mangroves can be related to their occurrence along the land-sea interface (planning has largely focused on marine ecosystems) and to their relatively extensive distribution, making their frequently represented areas small in percentage terms.

Figure 3.5. Representation of geomorphology classes and ecosystems within different levels of overlap of priority conservation areas. Graphs show the percentages of the total area of (a) marine geomorphology classes and subclasses and (b) selected coastal and marine ecosystems that correspond to three classes of frequency of spatial coincidence: low (1-2), medium (3-4), and high (5-6).

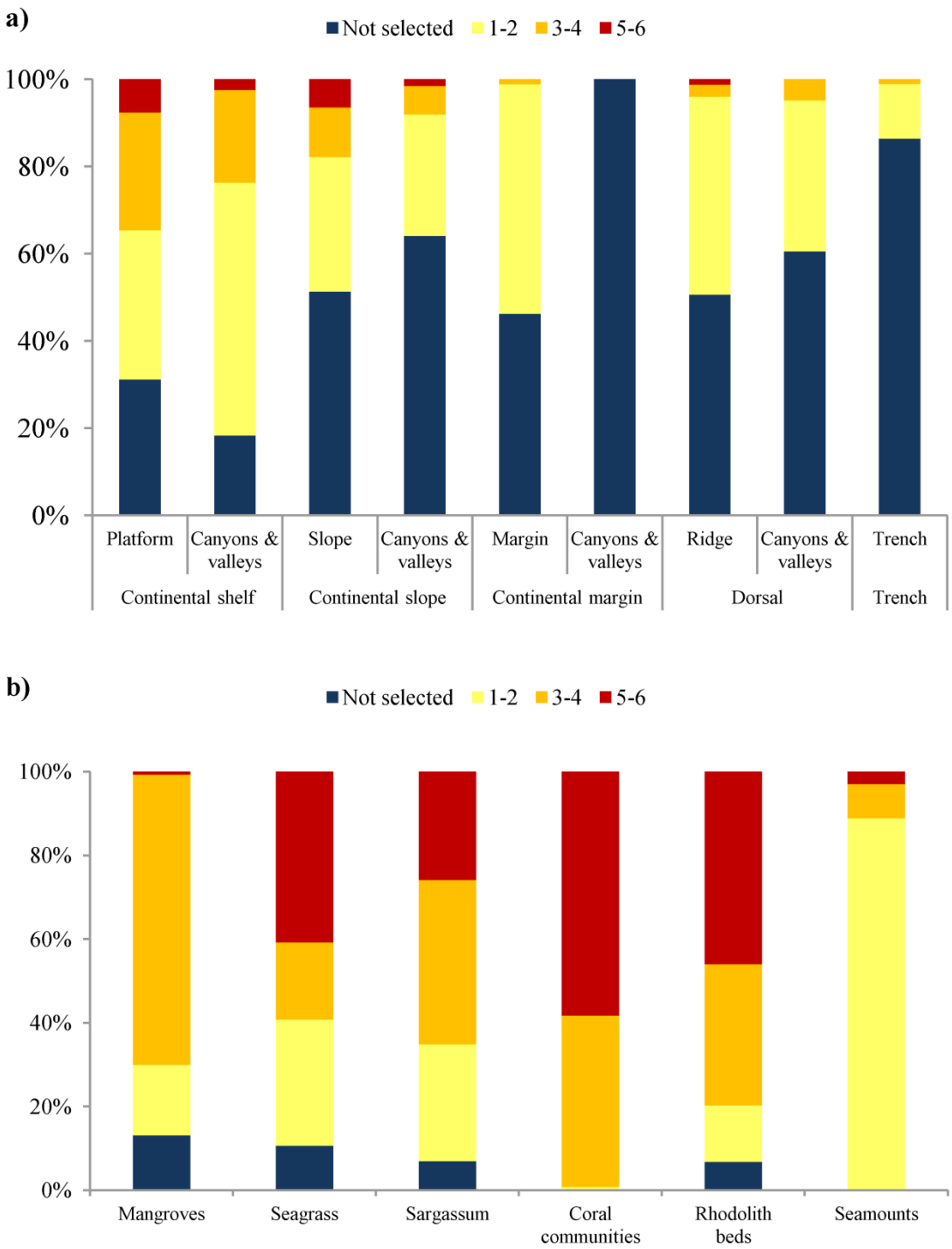

Figure 3.5. Representation of geomorphology classes and ecosystems within different levels of overlap of priority conservation areas

Achievement of goals, implementation of spatial actions, and benefits of planning

Through interviews and surveys, experts assessed the reported and implicit goals of the planning exercises in the Gulf of California, and the extent to which these had been achieved (**Figure 3.6**). An explicit goal cited by all experts was identifying generic priority areas for conservation, while identifying the particular conservation actions required within these areas was generally considered to be implicit. Other explicit goals indicated by most experts were to document and assess biodiversity and threats and/or conflicts with human activities, as well as to guide conservation/resource use planning. The majority of experts agreed that, overall, reported goals had been achieved satisfactorily. Some experts disagreed with this perception, mostly because the application of planning outputs had progressed very slowly beyond local ongoing initiatives that lacked the perspective potentially provided by the planning exercises. Opinions regarding implicit goals were more varied. The use of existing priority maps by later plans was commonly perceived as a positive outcome from planning, but never reported or mentioned as an implicit goal.

Figure 3.6. Expert opinion on reported and implicit planning goals. (a) Percentages of experts (N = 13) that identified the described planning goals as being explicitly reported (black portion of the bar) or implicitly considered (grey portion) by planners when developing the planning exercises; only the last goal was common to all planning exercises (either implicitly or explicitly). (b) Percentages of experts that considered the overall achievement of goals within each level of achievement, from "Not at all" (experts strongly disagreed that goals were achieved) to "Fully" (experts completely agreed that overall goals were met).

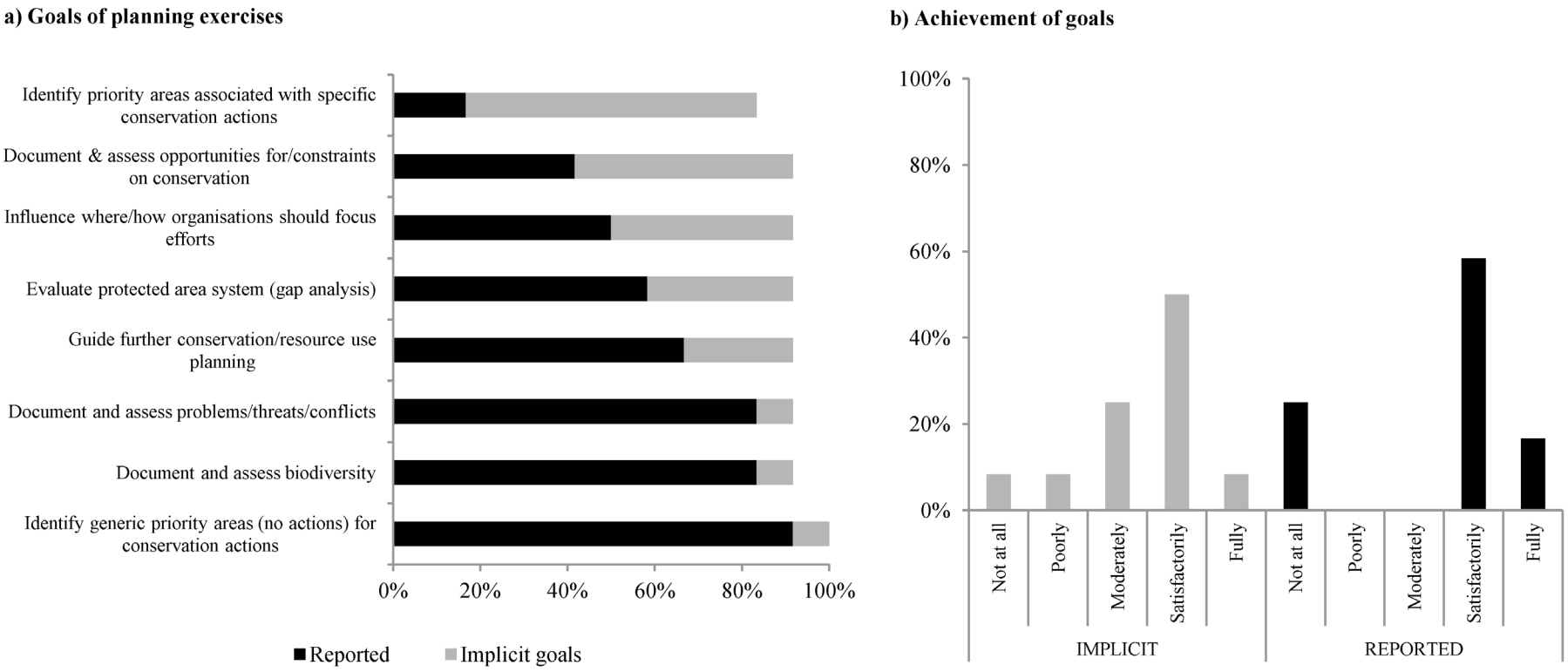

Figure 3.6. Expert opinion on reported and implicit planning goals

Despite differences in opinions, experts commonly indicated that plans had guided or in some way influenced the implementation of spatial actions within priority areas (**Figure 3.7**). According to more than half of experts interviewed, a common outcome of plans was the implementation of multiple-use protected areas (e.g., Bahia de los Angeles, Espiritu Santo National Park, San Lorenzo Archipelago National Park, San Pedro Martir Biosphere Reserve). Two actions, although only cited a few times, were the creation of no-take zones (e.g., Guaymas Hydrothermal Vents) and the promotion of socio-ecological monitoring or research projects within priority areas (e.g., PANGAS: http://pangas.arizona.edu/es/inicio). To a lesser extent, experts also considered that plans had contributed to developing coastal-marine development and zoning plans, mainly by providing information to agencies and industry regarding the location of areas of conservation value.

Figure 3.7. Expert opinion on how planning influenced or led to spatial implementation of actions. Bars denote the percentages of experts (N = 13) that considered the described actions as being implemented as a result of planning, either directly or indirectly.

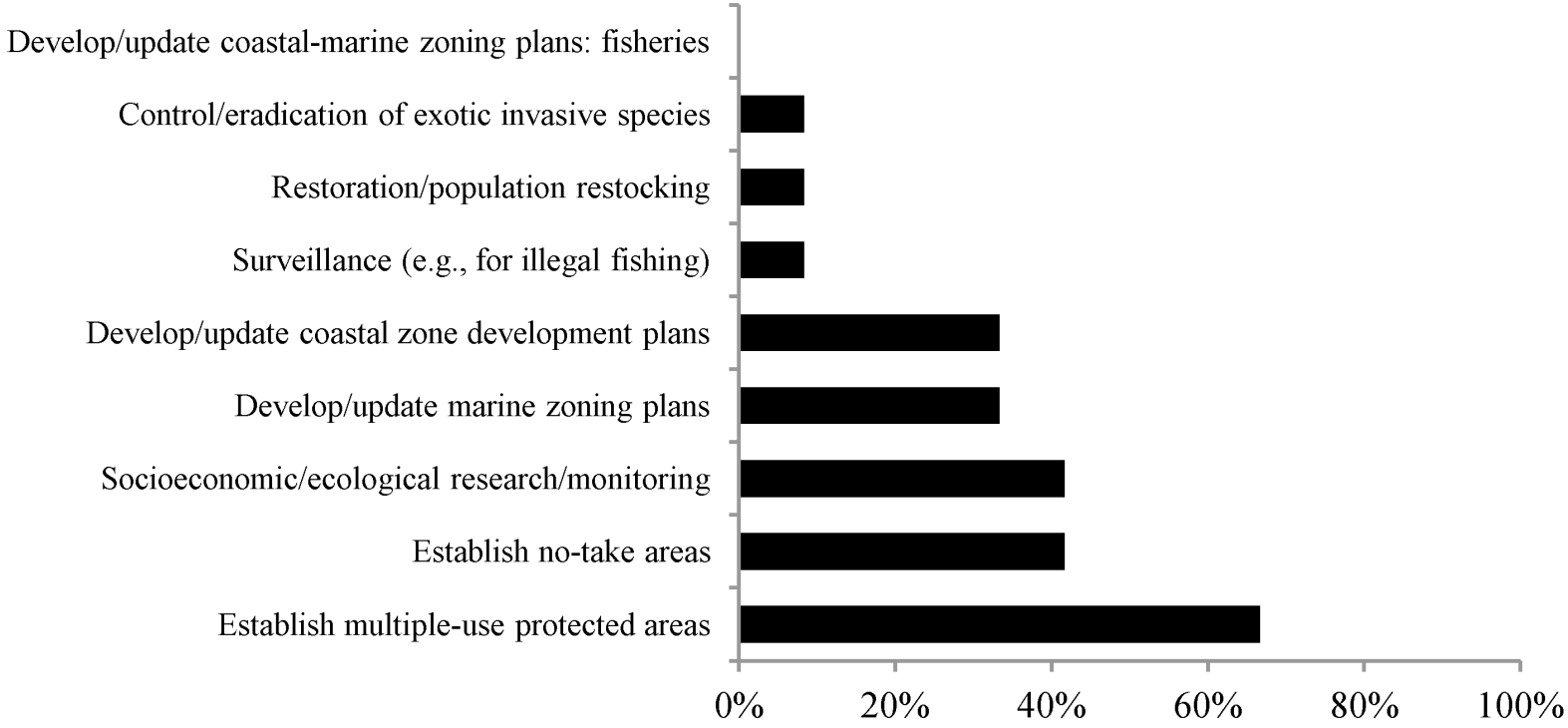

Figure 3.7. Expert opinion on how planning influenced or led to spatial implementation of actions

According to experts, factors that facilitated spatial implementation included: bringing priority areas to the attention of agencies, organizations, and general public; having a regional or national spatial framework that provided sound scientific support for ongoing marine conservation efforts and supported decisions by agencies regarding the creation or zoning of MPAs; and participation of multiple stakeholders in planning, in particular those directly influencing conservation decisions in the region. On the other hand, some factors were perceived as major constraints on the implementation of plans or causes of the slow uptake of plans in general, including: conflicting mandates or priorities between conservation and natural resource management agencies and organizations; limited technical capacities and public conservation resources (particularly at the municipal and State levels); insufficient political determination (e.g., from State governments) to implement MPAs; the need to further refine the design of conservation areas (e.g., incorporate socioeconomic constraints, biological connectivity, zones) and schedule their implementation; and the fact that plans have not been incorporated by relevant agencies into their strategic plans and annual working programs, particularly agencies not directly related to conservation, e.g., tourism, water management, infrastructure, and development.

In addition to spatial actions, experts considered that planning brought other benefits within the region (**Figure 3.8**). Despite differences in opinions, institutional and knowledge/skills were commonly perceived as benefits. Institutional benefits included: standardizing or having more rigorous planning processes, facilitating the development/implementation of regional or local conservation strategies, and influencing conservation strategies and actions by organizations/agencies/partners, guided by the priority maps. Other important benefits identified were developing skills/expertise of organizations in conservation planning and building the supporting scientific evidence for decision-making by participating organizations, agencies or partners. Among the social benefits, improving collaboration and coordination between organizations, creating a regional conservation approach for conservation, and building trust among organizations were apparent common outcomes. In comparison, direct financial benefits (e.g., making better use of available conservation resources, improving efficiency in the operations of organizations) were less commonly noted.

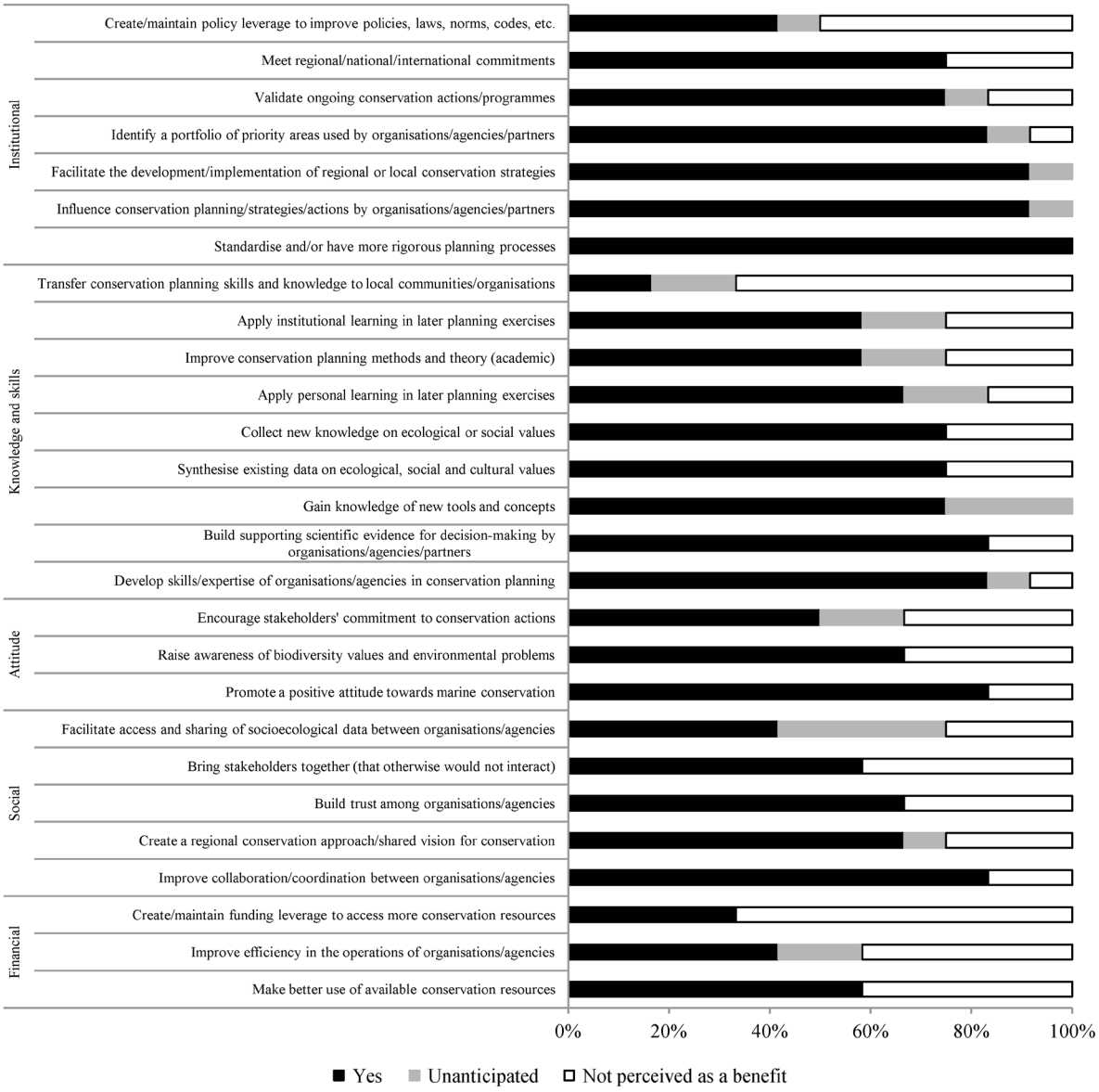

Figure 3.8. Expert opinion on the benefits of planning apart from spatial implementation of actions.

The black portion of each bar represents the percentages of experts (N = 13) that considered the described benefits were a result of planning; the grey portion depicts the percentages that indicated that those benefits were unexpected; white represents the percentages of experts not citing those benefits as outcomes of planning.

Discussion

The state of marine conservation planning in the Gulf of California

Despite the growing number of studies and marine applications of SCP methods and tools, uptake has been relatively limited in the Gulf of California. Only two planning exercises (Sala et al. 2002; Ulloa et al. 2006) used conservation planning software to find least-cost solutions that achieved conservation objectives, even though these tools have been readily available for more than 10 years (Ball et al. 2009), with applications to marine planning dating back to 1999 (Leslie 2005). Emerging issues, including targeting of ecosystem services and consideration of land-sea connections, have also received limited attention in the Gulf. Notably, none of the planning exercises dealt with climate change or scheduling of conservation actions.

Conservation plans in the Gulf of California followed some best-practices regarding stakeholder involvement but fell short for others. Stakeholder input was decisive in five stages of planning: identifying broad goals; setting objectives; defining selection criteria for priority areas; supplying, validating, and generating spatial data to represent conservation features; and identifying priority areas (see Pomeroy & Douvere 2008 for a review on stakeholder participation in marine planning). Expert workshops facilitated stakeholder involvement in decision-making processes in the Gulf of California and contributed to filling gaps concerning ecological and socioeconomic data (q.v., Wheeler et al. 2008). In addition, highly participatory processes (e.g., SPM and CSGC) also contributed to building common visions of conservation problems and opportunities. In expert-driven processes, tools other than conservation planning software helped to visualise spatial data (CONABIO 2007a; Enr²quez-Andrade et al. 2005), facilitate group decision-making (Arriaga-Cabrera et al. 1998), and delineate priority areas (Morgan et al. 2005). A notable gap in most exercises was the limited input from natural resource users and industry, which was mostly restricted to providing information on highly used areas. Including all stakeholders, but particularly resource users, from the outset of planning helps to build ownership and support (Gilliland & Laffoley 2008; Lundquist & Granek 2005), and has been pivotal in successful regional-scale exercises in marine planning (Fernandes et al. 2005; Gleason et al. 2010).

Only two planning exercises set quantitative objectives (Sala et al. 2002; Ulloa et al. 2006), which contributed to more explicit, transparent, and justifiable decision-making processes (Margules & Pressey 2000). While qualitative objectives (e.g., maximise species richness) were common to all exercises, only Sala et al. (2002) quantified and targeted species richness (for fish species). Both exercises that used quantitative objectives also partitioned these objectives between occurrences of the same features in different bioregions. Incorporating bioregional representativeness as a conservation objective is critical to ensuring that MPA networks capture the variety of biological assemblages, including species that are poorly recorded or undescribed (Roberts et al. 2003a). Bioregional representation has been fundamental to other marine planning initiatives, e.g., the California Channel Islands Marine Reserves (Airame et al. 2003) and the Great Barrier Reef Marine Park rezoning (Fernandes et al. 2005).

Incorporating spatially explicit socioeconomic data (e.g., human activities, opportunity costs associated with use restrictions, and costs of managing MPAs) in marine planning is critical to minimizing the costs of interventions (Ban & Klein 2009; Klein et al. 2008a). Exercises from the Gulf of California illustrate contrasting approaches to incorporating threats and costs in planning. The two exercises that used conservation planning software explicitly avoided threatened areas and minimised conflict with fisheries (Sala et al. 2002; Ulloa et al. 2006). In both cases, field data on fishing effort informed the cost layers. Despite potential limitations and coarse spatial resolution of these data, they are likely to be more accurate than surrogates for socioeconomic variables (Weeks et al. 2010b). In contrast, the B2B exercise (Morgan et al. 2005) focused on areas where conservation features of interest and threats intersected (i.e., prioritised vulnerable areas), an approach that aims to reduce further loss of conservation features. Although these approaches might seem opposite, they are likely to converge. Application of actions following selections of areas such as those by Sala et al. (2002) and Ulloa et al. (2006) would require scheduling, in which areas of both high irreplaceability and high threat would need to be prioritised (Pressey & Bottrill 2008). Surprisingly, most exercises identified priority areas independently of their vulnerability or level of use, either assessing threats independently (Enr²quez-Andrade et al. 2005) or designing strategies to address threats after priorities were set (Arriaga-Cabrera et al. 1998; CONABIO 2007a; Enr²quez-Andrade & Danemann 1998).

An interesting finding of the CSGC exercise (Enr²quez-Andrade et al. 2005) was that priority areas tended to be heavily used or highly threatened, indicating that conservation management in some used and threatened areas might be inevitable. This emphasises the need for better data on threats and costs and an understanding of management actions that are feasible and effective in contested parts of the Gulf. Conflict over conservation and extractive uses can be reduced, but not altogether avoided, by explicitly incorporating costs and threats in the prioritization process. Combining multiple costs (e.g., to different stakeholder groups) is challenging for marine planners (Ban & Klein 2009), but should be considered in the Gulf of California.

Despite spatial data limitations, ecological processes (e.g., migrations and oceanographic phenomena associated with high productivity) were considered by most planning exercises. Readily-available biological and oceanographic data (e.g., satellite-derived data on spatial variability of high-productivity areas) (Hidalgo-Gonz§lez & Alvarez-Borrego 2011), combined with existing oceanographic-ecological models (Cudney-Bueno et al. 2009; Marinone et al. 2008), can help to improve planning for processes in the Gulf of California and elsewhere. Recent spatial planning that could be adapted to the Gulf has incorporated extensive pelagic and oceanographic phenomena (Grantham et al. 2011) and considered the effects of climate change (Hobday 2011). These approaches will help to ensure local functionality of priority areas as well as contributing to regional-scale conservation objectives (Roberts et al. 2003b), including those related to fisheries.

Notwithstanding the global importance and degradation of the services provided by coastal marine ecosystems (Barbier et al. 2010; Worm et al. 2006), particularly in the Gulf of California (Aburto-Oropeza et al. 2008; Ezcurra et al. 2009), these were not considered explicitly by any of the planning exercises. This gap might have been partially addressed, however, by targeting conservation features based on their potential economic value, including: areas of high productivity and plankton abundance for fisheries (Arriaga-Cabrera et al. 1998; CONABIO 2007a); coastal ecosystems (e.g., coral reefs and kelp forest) that provide protection to sandy beaches against wave action; estuaries and marshes as exporters of nutrients/energy and buffers against land-based threats (Enr²quez-Andrade & Danemann 1998; Morgan et al. 2005); and mangroves as nurseries for commercial fish and invertebrates (Enr²quez-Andrade et al.

2005; Sala et al. 2002). The omission of ecosystem services is common in marine spatial planning (Granek et al. 2010), although there have been recent advances in their valuation and mapping (Silvestri & Kershaw 2010; Tallis & Polasky 2009). Further research is needed for marine applications (Chan & Ruckelshaus 2010).

The importance of land-sea interactions was recognised by most planning exercises, both in terms of cross-system threats and ecological connections. However, spatial data on, or models of, land-based threats were very limited and only used in a couple of plans. Prioritisation of catchments to mitigate land-based threats was not considered by any plan. Considering land-based threats can alter marine priority areas (e.g., locating them away from imminent land-based sources of pollution when possible) (Tallis et al. 2008) but can also help to integrate terrestrial and marine planning and facilitate marine management (Klein et al. 2010). Only one exercise (Enr²quez-Andrade et al. 2005) explicitly targeted freshwater and terrestrial areas important to maintaining ecological processes connecting land and sea. Adjustments of priority maps can also be required to plan for land-sea processes, e.g., species migrations between disjunct terrestrial and marine habitats (Hazlitt et al. 2010) or input of marine-derived nutrients into coastal and insular ecosystems (Lombard et al. 2007). The lack of consideration of land-sea connections in the National Protected Area System of Mexico was emphasised by Ortiz-Lozano et al. (2009), and highlights the need to improve data and models on spatial and temporal occurrence of land-sea processes and cross-system threats for marine planning (Alvarez-Romero et al. 2011).

Convergence and divergence between plans

Based on a spatial overlay of priority maps, Aburto-Oropeza and Lopez-Sagastegui (2006) noted important spatial coincidences between the outputs of different marine planning exercises in the Gulf. These similarities have led to a general perception among conservation practitioners in the region that planning exercises identified the same priority areas. My analysis shows that, despite areas of convergence, there are important differences, some of which can be related to the planning approach, methods and extent. The small overlap between all exercises raises the question of whether the convergence of multiple independent plans is in fact pointing to areas that hold valuable biodiversity features and processes that have been recognized widely by planners. In

some cases, areas commonly identified as marine priorities seem to be located in better studied areas (e.g., Upper Gulf, Midriff Islands, Loreto, La Paz Bay), a common outcome in marine planning in general (Ardron et al. 2008) and a plausible explanation of why these areas also overlap with existing MPAs (Weeks et al. 2010a), some of which were established prior to prioritization exercises.

The similarities and differences in plans might be explained by commonalities between them, including: similar data; similar approaches (e.g., expert-based vs. data driven, prioritization of threatened areas with biodiversity value, focus on nearshore environments); the use of priority maps from previous exercises; and the resolution of data and targeted conservation features. Contrary to expectations, clustering based on spatial similarity did not show clear differences between priority maps developed at different spatial scales**.** An important result from the Kappa location analysis was the greater similarity values for all exercises. This indicates that the relative proportions/size of priority areas (which can be related to the extent of planning domain) can be an important driver of overall spatial similarity between overlapping priority maps.

Interestingly, grouping based on spatial similarity did not closely match that based on planning objectives and datasets. Divergence between the methodological and spatial similarity phenograms supports the idea that various elements, in addition to data and objectives, played an important role in defining the distribution of conservation priorities. These could include inaccuracy in manual delineation of priority areas from expert-based exercises, incorporation of updated datasets, and random error. Unfortunately, the limited and fragmented spatial data available from these studies did not allow further investigation of these differences.

Size and perimeter have important design implications, both biologically and practically (Leslie et al. 2003; Roberts et al. 2001b). Remarkably, priority areas proposed by the only exercise that explicitly included minimum size as a design criterion (Sala et al. 2002) were significantly smaller than those from other exercises (**Figure 2a**). In part, this is explained by the reduced number of planning units, but also reflects differences in objectives and how these can influence priority area design.

Convergence of priorities was observed in well studied shallow and benthic marine ecosystems (e.g., coral reefs, seagrass, *Sargassum* beds). Coastal ecosystems were also represented in established MPAs, some of which were established at least partially in response to the exercises reviewed here. The recent addition of an MPA in the Gulf of California to protect the benthic communities (hydrothermal vents) in the Guaymas Basin ï an area identified as a marine priority by only one exercise ï is an exception to the general trend. The under-representation of deep-ocean and pelagic environments in the Gulf of California, in comparison with nearshore/shallow ecosystems, reflects the relatively limited data for deep-ocean and pelagic environments in general (Glover & Smith 2003; Webb et al. 2010) and the emphasis in the protection of near-shore ecosystems (Spalding et al. 2010; Wood et al. 2008). Among the solutions, vertical zoning provides a way of designing priority areas suitable for protecting pelagic and/or benthic environments as required (Grober-Dunsmore et al. 2008). New technologies are allowing the exploration of deeper environments in the Gulf of California (Aburto-Oropeza et al. 2010) and the data from these studies should be used to update marine planning exercises.

Uptake of planning: implementation of spatial actions and other benefits

Assessing the implementation of plans was not an objective of this study. However, based on the information provided by experts, the implementation of new MPAs in the region has been influenced by some of these planning exercises. It seems that monitoring and research activities have resulted from the long history of prioritizing conservation areas in the region. However, as the experts noted, enforcement and many other problems (Rife et al. In review) indicate that MPAs are not necessarily fulfilling the objectives stated in their decrees.

An element regarded by some experts as a constraint on the implementation of spatial actions was the lack of regard for ongoing refinement of priority area designs to maximise the likelihood of implementation. Such refinements include defining simple and readily recognizable boundaries to maximise compliance and facilitate enforcement. In other regions, implementation has been a critical criterion in planning exercises that have led to successful application of the planning outputs (Fernandes et al. 2009; Fernandes et al. 2005; Lombard et al. 2007). These exercises also illustrate the need to

work with stakeholders once preliminary priority maps are proposed. The execution of specific actions such as surveillance, species recovery, and invasive species eradication/control programs within priority areas were seldom mentioned by experts as implemented following planning. This suggests that the failure to identify specific actions within priority areas (to address problems associated with particular areas) may also be limiting the value of conservation plans. Interestingly, none of the interviewees mentioned actions related to fisheries management, which could in part reflect the limited participation of the fisheries sector in planning and the general disconnect between this sector with the national/regional conservation agenda.

The implementation of MPAs in the Gulf of California, as everywhere in Mexico (Bezaury-Creel 2005), has been mostly opportunistic, thus limiting the uptake of marine planning outputs. Reasons underlying this approach include incomplete and biased socioecological data, accelerated degradation of certain areas (and reactive local efforts to protect them), and concurrence of suitable socioeconomic and political conditions. These elements can, however, be considered and incorporated into SCP initiatives by following an adaptive approach. Adaptive conservation planning is critical to make the best use available socioecological data, deal with ongoing loss of biodiversity, and take advantage of emerging conservation opportunities (Grantham et al. 2009). Some key requirements to adaptive planning are: setting explicit and preferably quantitative conservation objectives (critical to the transparency, traceability and revision of planning decisions); developing spatial data infrastructures (needed to organise and share data to revise the plans or to inform other plans, as well as to continuously and systematically update socioecological data derived from monitoring and research); and assessing the effectiveness or response of socioecological systems to conservation interventions (essential to adjust objectives, priority area design, and implementation strategies). Unfortunately, few long-term monitoring programs (a pre-requisite to adaptive management) exist in the region today, and there are no explicit intentions to update the plans, even when information from these programs is available. The lack of these two strategies is a major limitation of marine planning in the Gulf of California. Conservation planning initiatives in South Africa exemplify how national and regional-scale plans are being documented, updated and implemented under a national multi-scale planning framework (e.g., South African National Biodiversity GIS: http://cpu.uwc.ac.za/). Additional elements that have constrained or facilitated

implementation include institutional continuity, funding availability, and political trends, so these need to be considered by planners.

Conclusions

The Gulf of California has seen seven marine planning exercises in the past decade. Yet uptake of marine applications of SCP methods and tools has been relatively limited, and the existence of multiple plans within the region pointed to some difficulties as well as benefits. On one hand, multiple plans have contributed to uncertainty, confusion and tension between stakeholders. Multiple plans may have also contributed to the spreading and inefficient use of financial and human resources, as well as to unnecessary redundancy in actions. Despite good intentions, the coexistence of multiple plans (particularly when these are developed without considering pre-existing plans) can hamper the goal to create a common approach and focus for conservation, thus resulting in uncoordinated actions undertaken by the different stakeholders. On the other hand, the existence of multiple planning processes and plans seems to have contributed to improved methods, data and tools, as well as being the basis for more robust and refined plans developed later.

Given that the National Marine Gap Analysis (CONABIO 2007a) considered the outputs of previous plans (incorporating some priority areas delineated in previous exercises) and engaged with stakeholders who participated in previous exercises, this plan has now been widely accepted and is considered (at least by environmental organizations and agencies) as the national framework for marine conservation in Mexico, including the Gulf of California. Thus, the logical next step is to build on this plan and refine priority areas (which provide the required regional context) into conservation areas associated with specific actions (e.g., through zoning) appropriate to deal with current threats, incorporating local socioeconomic data and ensuring participation of local stakeholders. This refinement should include integration with terrestrial and freshwater priorities, in particular for areas where important land-sea connections have been identified (Enr²quez-Andrade et al. 2005) or where land-based threats are a concern. A stronger commitment of agencies from different sectors (environment, fishing, tourism, water management, infrastructure and development) is

also needed to incorporate planning results into the relevant policies, regulations and annual programs, as well as to continue participating in the refinement of plans. Increased coordination between organizations (from local to international) will be fundamental to implement a functional network of MPAs that contributes to achieving conservation goals and the sustainable use of natural resources in the Gulf of California.

Chapter 4. Modelling catchment nutrients and sediment loads to inform regional management of water quality in coastal-marine ecosystems: a comparison of two approaches

Abstract

Human-induced changes in flows of water, nutrients and sediments have downstream impacts on estuarine and marine ecosystems. An urgent need has been recognised to study and quantify these impacts and to systematically undertake mitigation or remediation actions for many areas around the World. Modelling nutrient and sediment loads and contributions from subcatchments can help to prioritise management and protection interventions to mitigate the impacts of altered flows of nutrients and sediments on ecosystems. Catchment modelling approaches to identify sources of nutrients and sediments vary in data requirements, outputs, and applications. I compared two models (N-SPECT and SedNet) commonly used to prioritise areas for management of water quality in coastal-marine ecosystems. I assessed their application to the Gulf of California, Mexico, a global marine biodiversity hotspot where major land-based threats to marine habitats have been documented. I found important differences in performance between models. Overall, SedNet performed better than N-SPECT. SedNet consistently estimated spatial variations in runoff with higher accuracy and, unlike N-SPECT, its modelled total suspended sediment (TSS) loads were mostly within the range of uncertainty in observed loads. Data requirements were not an important limitation on the implementation of either models in the study area, but the time required to collate data was significant. Outputs from both models were appropriate to represent broad spatial patterns of pollutant supply across the study area, but spatial differences in model outputs have implications for marine conservation planning, including their applicability to river plume modelling and environmental risk assessments.

Introduction

Human-induced changes in flows of nutrients, pesticides, sediments, and fresh water are major threats to coastal and marine ecosystems worldwide (Doney 2010; Tilman et al. 2001). Nutrient enrichment of coastal and marine waters associated with agricultural fertilisers creates eutrophic and hypoxic or anoxic conditions that affect the functioning of marine ecosystems and the status of biodiversity and human health (Diaz & Rosenberg 2008; Gabric & Bell 1993; Howarth 2008). Pesticides from agriculture negatively affect sensitive habitats and species such as mangroves, seagrass, marine mammals, seabirds, and various fish and invertebrates (Duke et al. 2005; Haynes et al. 2007; Readman et al. 1992). Loads of sediments derived from land clearing, urbanisation and agriculture have significantly increased in many regions (Walling 2006) and are a major threat to vulnerable ecosystems such as coral reefs and seagrass beds (Cabaco et al. 2008; Maina et al. 2011; Rogers 1990). In contrast, damming of rivers reduces the delivery of river-borne sediment and nutrients to the sea (Walling 2006), but also with potential adverse effects, significantly altering biogeochemical cycles and species composition in estuarine and marine ecosystems (Humborg et al. 2000). Managing such land-based threats is increasingly recognised as a crucial component of maintaining healthy coastal and marine systems globally (Cherubin et al. 2008; Halpern et al. 2009; Maina et al. 2011).

Regional studies to identify and assess land-based threats to coastal-marine ecosystems and to explore options to prevent or mitigate these threats are urgently needed (Burke & Sugg 2006; McKergow et al. 2005b). Catchment models are powerful tools to explore management options to improve water quality (McKergow et al. 2005a, b; Walling et al. 2011) because they link sources of pollutants (e.g., sediment, nutrients, pesticides) to affected areas and ecosystems. Furthermore, they facilitate identification of dominant processes associated with production and delivery of pollutants (Drewry et al. 2006; Walling et al. 2011), and thus help to determine appropriate strategies to minimise downstream impacts. Applications of catchment modelling for water quality control include identifying erosion hotspots and estimating pollutant loads delivered to the sea, which can then be incorporated into river plume models to assess vulnerability of coastal-marine ecosystems to land-based threats (Burke & Sugg 2006; Devlin et al.

2012b; Maughan & Brodie 2009). Mapping major sources of pollutants within catchments is thus needed to guide management to prevent or reduce downstream impacts (Brodie et al. 2009). Management actions include implementation of best practices in agriculture and livestock management, water treatment, restoration of riparian vegetation, protection of erosion-prone areas, and gully stabilisation (Bracmort et al. 2004; Chang et al. 2012; Thorburn et al. 2011; Wilkinson & Brodie 2011).

Different modelling approaches have been used to estimate the supply and delivery of sediment and nutrient loads, varying in complexity, data requirements and intended spatial and temporal scales of application. Many modelling approaches have been developed for local, field-scale applications (Jetten et al. 1999), including event-based or daily time-step models that are appropriate for short-term and local management or ongoing water quality monitoring (e.g., SWAT: Gassman et al. 2007). These models, however, are generally too data-intensive and detailed to be applicable for large or multiple catchments, and their timeframes too short to inform long-term regional water quality management. Fewer models appropriate for large-scale applications exist, although several regional catchment models are pertinent (e.g., N-SPECT, SedNet, SWIM). For modelling approaches to be relevant for identifying and assessing land-based threats to coastal-marine ecosystems, several considerations are important: intended spatial and temporal scales, data availability *vs.* data requirements of models (climatic, flow, *in situ* water quality sampling), existing technical expertise, and time and budget constraints (Walling et al. 2011). Commonly, flow data and water quality sampling or monitoring are spatially incomplete (covering only some parts of regions) and temporally fragmented (spanning limited and/or different time periods). Data gaps, together with limited technical capacity, have restricted regional applications of more complex models applicable to regional scales.

The aim of this study is to compare two prominent regional catchment models (and N-SPECT: NOAA 2008; SedNet: Wilkinson et al. 2004) in terms of their performance, data requirements, ease of implementation and utility for regional-scale planning. Both models have been used to estimate end-of-river loads of sediment and nutrients and to identify areas for intervention upstream to minimise effects on marine ecosystems. Both models have been employed in different regions, including extensive applications (and N-SPECT - Mesoamerican Reef: Burke & Sugg 2006; McKergow et al. 2005a; e.g.,

SedNet - Great Barrier Reef: McKergow et al. 2005b) and small-scale exercises (e.g., Makaha Valley, Hawaii: Carter et al. 2005; and Aransas County, Texas: Dunton 2009). Both models are appropriate to estimate long-term annual pollutant supply and end-of-river loads (Eslinger et al. 2005; Wilkinson 2008) and are also considered to be good compromises in terms of model scope, data requirements and performance. Here I apply the models to catchments draining to the Gulf of California, Mexico, where there is an urgent need to study and incorporate land-based threats to marine ecosystems into conservation planning (Alvarez-Romero et al. In review). The study area also has highly variable hydrological and climatic conditions and limited data, making it a good test case for many similar regions around the world. Previous studies have examined nutrient enrichment of coastal waters in the Gulf of California, but focused on irrigated agriculture (Ahrens et al. 2008; Riley et al. 2001), and local scales (Christensen et al. 2006). Our study investigates the use of catchment models to estimate runoff by analysing whole catchments and the effects of several land uses.

Methods

Study area

The Gulf of California is a marine ecosystem globally recognised for its rich and unique sea life (Olson & Dinerstein 2002), and is currently threatened by sea- and land-based activities (Lluch-Cota et al. 2007). Human population density is relatively low in the catchments draining into the Gulf of California, but is rapidly increasing, with associated increases in threats to the marine environment (Lluch-Cota et al. 2007). While the western coast remains comparatively undisturbed, many eastern coastal areas are affected by land-based threats, including agriculture, urbanisation, aquaculture, and damming (Lluch-Cota et al. 2007; P§ez-Osuna & Ruiz-Fern§ndez 2005). Although the effects of land-based pollution are still poorly understood, the potential impacts can extend hundreds of kilometres from the coast (Beman et al. 2005). Intensive use of pesticides in agricultural areas is a particular concern (Galindo-Reyes et al. 1999; Gutierrez-Galindo et al. 1992; Ni¶o-Torres et al. 2010). Alteration of freshwater and sediment inputs from the Colorado River has already caused major changes in the

Upper Gulf, altering the habitat for commercial and endangered species (All 2006; Carriquiry & S§nchez 1999; Rowell et al. 2008).

The study focuses on selected catchments draining into the Gulf of California, including one of the most important agricultural regions in Mexico, the Yaqui Valley. The region stands out globally for the intensive use of agrochemicals and fertiliser (Ahrens et al. 2008; Naylor et al. 2001). Effects of terrestrial runoff (e.g., associated with the intensive use of fertilisers) have been observed in offshore marine areas within the Gulf (Beman et al. 2005). This is particularly relevant because multiple adjacent coastal-marine areas are considered national, regional and/or provincial conservation priorities (CONABIO-CONANP-TNC-PRONATURA 2007; SEMARNAT 2006). I selected these catchments (**Figure 4.1**) for several reasons: they include all vegetation types found within the catchments draining into the Gulf, which is important if a Gulf-wide study is to be undertaken later; they comprise a variety of catchment sizes, from very large (e.g., Yaqui and Mayo: 67,629 and 13,303 km^2 respectively) to small coastal catchments (~50 km^2), thus exemplifying the variety of transport and depositional processes associated with catchment size (Prosser et al. 2001); and there is a marked climatic variation and limited stream gauge data (with dryer areas being poorly recorded)across the region, providing a test of the modelsô capability to represent variations in runoff (Wilkinson et al. 2006). These characteristics make the region a good case study to test the application of these two catchment modelling approaches.

Figure 4.1. Key features of the study area relevant to catchment modelling: A) Geographic location; B) The selected catchments comprising the study area, all draining into the Gulf of California, also showing major reservoirs (used in SedNet to estimate changes in pollutant loads due to deposition of sediments and particulate nutrients), gauge stations, and conservation priority level of adjacent marine management units; C) The strong gradient in dryness across the study area, as illustrated by the ratio of potential evapotranspiration to precipitation (Eo/P) ratio; and D) Main land use classes depicting degree of human modification of the landscape (e.g., conversion of natural areas to cropland and pasture) potentially associated with spatial patterns of pollutant supply; the percentage of area occupied by each land use is indicated in brackets.

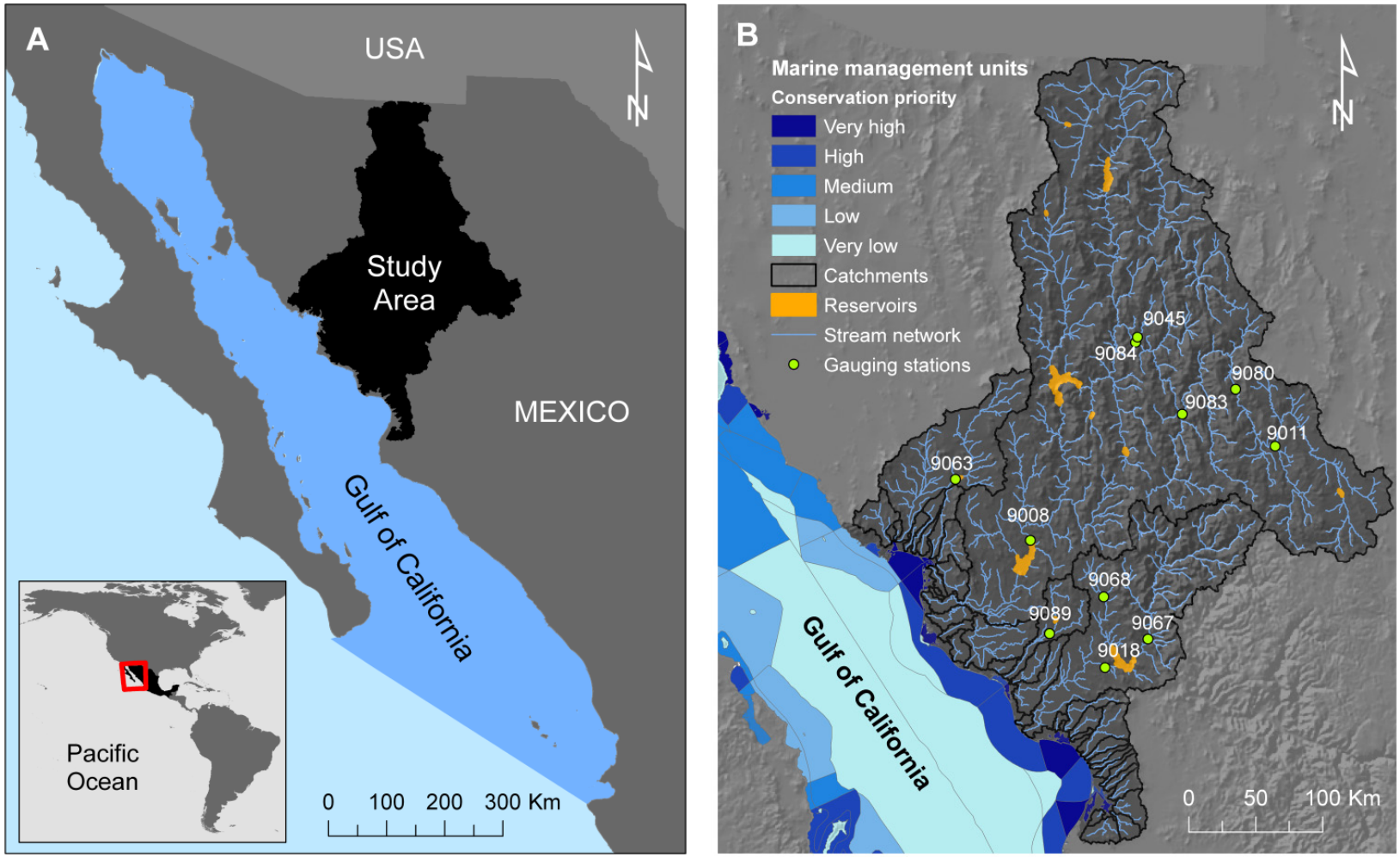

Figure 4.1. Key features of the study area relevant to catchment modelling

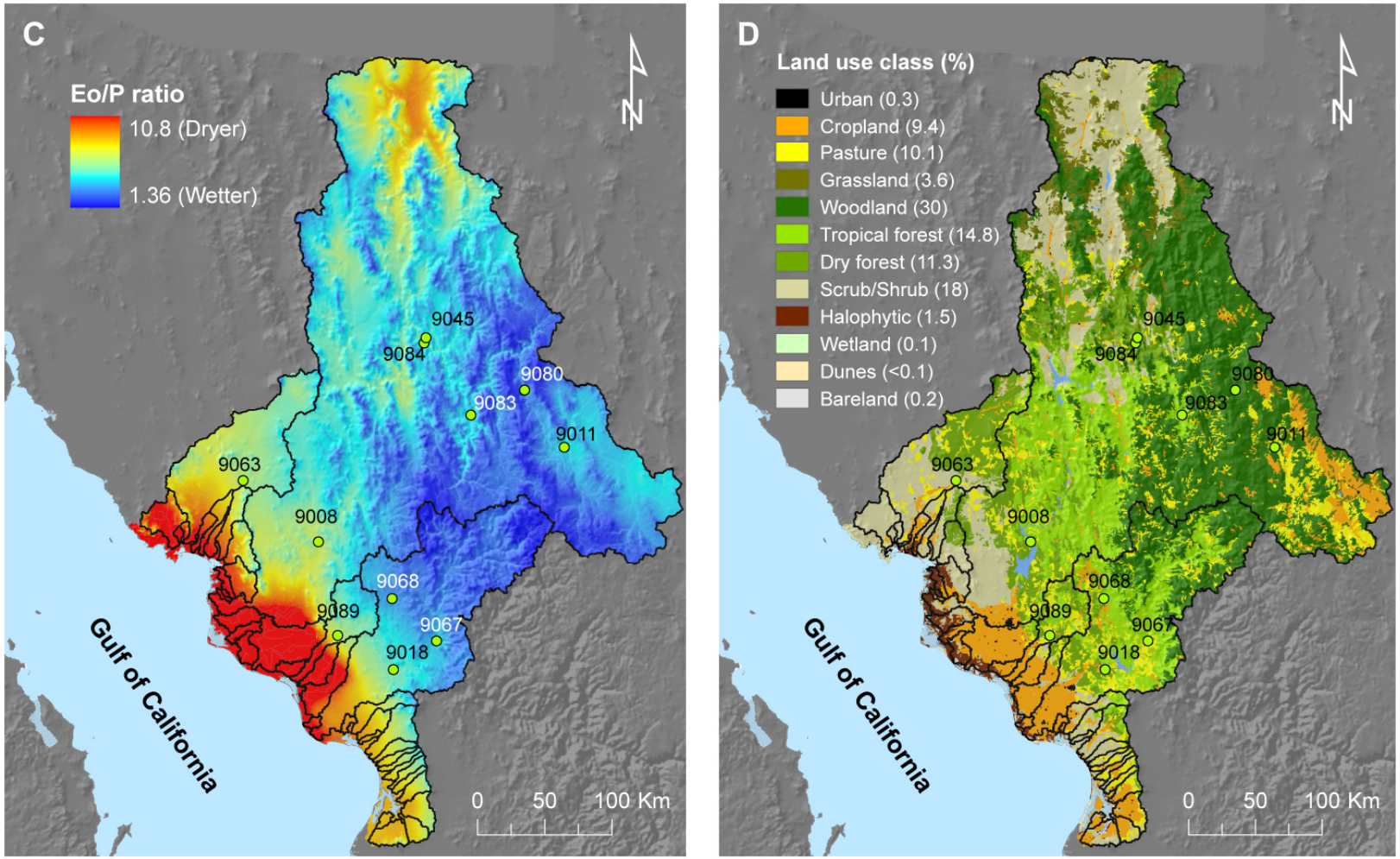

Figure 4.1. Key features of the study area relevant to catchment modelling (cont.)

Catchment model descriptions and data requirements

I compared two catchment models, N-SPECT and SedNet, which have common input datasets and similar outputs but differ in some important aspects (**Figure 4.2**). Both models rely on spatial datasets and process conceptualisations. SedNet outputs (including runoff, erosion and nutrient supply) are lumped within subcatchments. N-SPECT calculations are performed on a cell-by-cell (i.e., raster grid pixel) basis, and are hence limited by the coarse-resolution dataset used here. N-SPECT outputs therefore require additional software to summarise outputs for subcatchments.

Runoff calculations are one fundamental difference between N-SPECT and SedNet. Runoff estimates in N-SPECT are based on the USDA Soil Conservation Service Curve Number (CN) method, a widely used method for determining the changes in rainfall runoff based on the amount of precipitation and infiltration characteristics of land use and vegetation cover (hereafter land use), soil type, surface retention and impervious surfaces (USDA 1986). Although the method was originally designed for single-storm events and small catchments, it can be scaled to find average annual runoff values (NOAA 2008) and has been used in large-scale or regional studies (e.g., Burke & Sugg 2006; Dunton 2009; Paris & Cherubin 2008). In contrast, SedNet runoff calculations are based on regionalisation of flow parameters (mean annual flow and daily flow variability statistics) following least-squares regressions based on observed flow data from gauge stations and catchment characteristics, including annual rainfall and potential evapotranspiration (Wilkinson et al. 2006).

Both models estimate soil loss from hillslope erosion based on the Revised Universal Soil Loss Equation (RUSLE), which estimates annual soil loss based on a number of landscape characteristics (i.e., rainfall erosivity, soil erodibility, length-slope, and vegetation cover), which are considered to be the main drivers of soil loss (Renard et al. 1997). However, the sediment delivery ratio (SDR) applied by both models are quite different. While SedNet uses a constant value representing *a priori* knowledge of how erosion estimates relate to sediment delivery (e.g., 0.05), N-SPECT automatically creates a spatially distributed SDR based on drainage area (i.e., raster cell size), relief-length ratio (calculated based on a digital elevation model and flow direction grid) and

curve number, following Williams (1977). Additional erosion processes incorporated by SedNet, but not N-SPECT, include gully and river bank erosion.

Calculations of dissolved nutrient supply in both N-SPECT and SedNet follow the export or runoff coefficient approach (Johnes 1996), which is based on mean runoff concentration (commonly referred to as Event Mean Concentrations ï EMC, but see Bartley et al. (2011) for a discussion on differences in methods used to estimate these values) of particular pollutants and nutrients from different land uses. EMC coefficients are combined with an estimated runoff volume to calculate the pollutant supply. In addition, SedNet estimates the supply and export of particulate nutrients associated with hillslope, gully and bank erosion. For both sediment (suspended and bedload material) and particulate nutrients, SedNet also represents reservoirs and floodplains as sinks within the river network, to estimate the amount of material actually exported from the river network. Sinks (SedNet only) include loss of nutrients due to denitrification in reservoirs and lakes and accumulation or deposition of sediment in river floodplains and reservoirs and lakes, as well as deposition of in-channel bed material. N-SPECT does not consider depositional or accumulation processes and thus its results represent the supply of sediment and nutrients.

Figure 4.2. Major steps, inputs and outputs of N-SPECT and SedNet models. The left panel (grey area) depicts modelling steps in N-SPECT. The two columns on the right correspond to SedNet. Common steps are presented side by side and different box shapes and shading are used to describe the different elements (i.e., input data, parameters and outputs). Within each step, different font types and colours are used to indicate differences between the models (e.g., when input data, parameters or outputs are common to both models or exclusive to one).

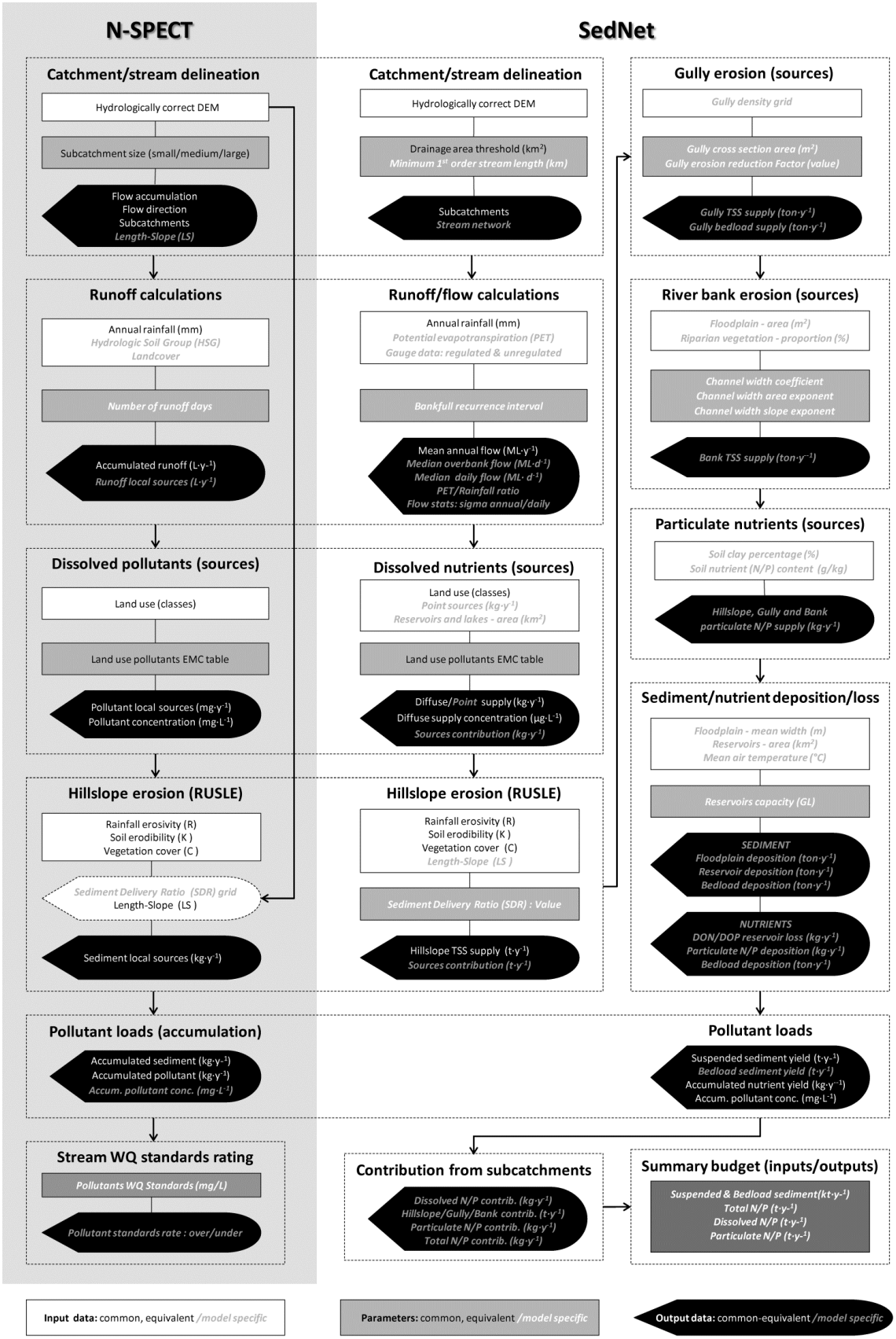

Figure 4.2. Major steps, inputs and outputs of N-SPECT and SedNet models

Model parameterisation

In this section, I describe parameterisation of models, including key data sources, processing and application for each major step (as described in **Figure 4.2**), indicating, when applicable, data limitations, adjustments, and whether these are used in both or only one model. A detailed description of data preparation and model settings are provided as **Supplementary Material**.

Catchment and stream network delineation

A digital elevation model (DEM) is used for delineating streams and subcatchments and routing of pollutant transport. I used a hydrologically corrected version of the 30 m resolution ASTER Global DEM (METI-NASA 2009), which was resampled using ArcMap Spatial Analyst (ESRI 2010) using bicubic interpolation to 240m. Similar resolutions have been used for other regional catchment modelling exercises (McKergow et al. 2005a) and provide a good balance between the extent of the study (approximately 600,000 km^2) and matching of resolution to that of other input data.

Both models delineate catchments automatically, based on user-defined parameters to set the subcatchment size. For N-SPECT I used the ósmallô subcatchment option, which resulted in 27 subcatchments with mean size of 3774 km^2 (range 385 to 8,652 km^2). For SedNet, I set the drainage area threshold at 40 km^2, which resulted in 1,655 subcatchments, with a mean size of ~60 km^2 (STD = 39 km^2). This ensured that most small coastal catchments were included in the SedNet study area. Differences between the stream network delineated using NSPECT and SedNet were minimal, hence I used the smaller subcatchments from SedNet to summarise the cell-based (i.e., local sources) results of N-SPECT.

Runoff and flow calculations

Annual precipitation (P) is used in both N-SPECT and SedNet to calculate runoff. I used the global high-resolution (~1km) annual precipitation grid (Hijmans et al. 2005) available online from WorldClim (http://worldclim.org/), which provides a good resolution for regional-scale catchment modelling (Heyman & Kobara 2006). The

number of rain days, required by N-SPECT, was calculated for each of three sections of the study area based on the curve number equation (USDA 1986). The model runoff was very sensitive to this parameter, and it was used to calibrate the model to observed (i.e., gauge data) runoff (**Supplementary material**).

The hydrologic soil group (HSG, i.e., the permeability or infiltration capacity of the soils) is used by N-SPECT, in combination with land use, to calculate curve numbers. I assigned HSG classes based on the classification of soil units proposed by CNA (1987) using a national map of soil units (INIFAP-CONABIO 1995). When two HSGs were assigned to a particular soil unit, the lowest permeability (maximum runoff) class was used, as suggested by Eslinger et al. (2005). Curve numbers (used in N-SPECT for runoff calculations) are determined by combining the HSG and land use. Higher curve number values are given for landscapes with more impervious cover (e.g., surface soils with high clay content or lands with sparser vegetation). I matched the land use classes of the study area with the Coastal Change Analysis Program (C-CAP) land cover classes (USDA 1986).

Potential evapotranspiration (PET) is used in SedNet to regionalise mean annual discharge. I used the Global Potential Evapotranspiration (Global-PET) dataset (available from: http://www.cgiar-csi.org), calculated with the Hargreaves evapotranspiration equation (Trabucco et al. 2008) using WorldClim datasets. For SedNet I also processed daily and monthly records from data from gauge stations obtained from the Mexican hydrometric database (BANDAS), available from http://www.imta.gob.mx. Of the 19 gauges within the study area, only 11 (**Figure 4.1**) had sufficient data (i.e., the minimum number of years recorded was 20 or 15 years for areas with no other stations available, and at least 10 years of these records should be uninterrupted) for the model.

Dissolved pollutants and nutrients

Land use is used to estimate the pollutant loads in both models and also to calculate potential runoff (N-SPECT only). I used the most updated national-scale (1:250,000) land use dataset available (**Figure 4.1D**), which corresponds to the year 2000

(SEMARNAT-UNAM 2002). This dataset was corrected for classification inconsistencies because different methods were used to create it (Vel§zquez et al. 2002).

I calculated pollutant event mean concentrations (EMC) based on a literature review on reported EMC for similar vegetation types, mostly restricted to North America, Australia, and South America. Values outside the commonly observed ranges in EMC for each land use were excluded. In particular, outlying values for total nitrogen and total phosphorus from forest land use in areas with high atmospheric deposition of nitrogen from industrial and high intensity agricultural areas were excluded (there was no indication of high deposition in the study area). Studies from North and South America (Lewis 2002; Lewis et al. 1999; Perakis & Hedin 2001; Perakis & Hedin 2002), Australia (Bartley et al. 2011; Brodie & Mitchell 2005; Harris 2001) and modelled data from the USA that accounted for atmospheric deposition (Smith et al. 2003) provided guidance to adjust EMC values for natural vegetation land use classes.

Hillslope erosion

Both models use RUSLE to estimate hillslope erosion based on rainfall erosivity, soil erodibility, length-slope factor, and land-use cover. Rainfall erosivity (R) was regionalised based on precipitation distribution and occurrence characteristics (duration, intensity, number of events) according to historical records available from climatological stations (q.v., Diodato & Bellocchi 2007; Mikhailova et al. 1997; Millward & Mersey 1999). I used a 1km annual precipitation grid (section 2.3.2) to estimate R based on the regionalised erosivity map and regression equations (**Table 4.1**) proposed by Cort®s-Torres (1991), as updated by Figueroa et al. (1991) and modified by SEMARNAT-UACH (2002). Six of the 14 erosivity regions described by Cort®s-Torres (1991) occur within the modelling domain.

Table 4.1. Regionalised regression equations used to calculate erosivity using annual precipitation (P). The numerals indicate the original codes assigned by Cort®s-Torres (1991) to the erosivity regions of Mexico.

Region	Equation
II	$3.45552\ I{:}P + 0.006470\ I{:}P^2$
IV	$2.89594\ I{:}P + 0.002983\ I{:}P^2$
VI	$6.68471\ I{:}P + 0.001680\ I{:}P^2$
V, VII	$0.71508\ I{:}P^{1.30751}$
X	$6.89375\ I{:}P + 0.000442\ I{:}P^2$

Table 4.1. Regionalised regression equations used to calculate erosivity

Soil erodibility (K) was calculated using the method proposed by FAO (1980), with soil data widely used for erosion studies in Mexico (SEMARNAT-UACH 2002) and other parts of the world (Heyman & Kobara 2006). For soil units with many horizons (i.e., soil layers with different characteristics) I calculated a weighted average of K for all horizons combined (FAO 1980; NOAA 2008).

The length-slope (LS) factor is calculated automatically in N-SPECT during the process of delineating catchments and stream networks using ArcMap Spatial Analyst (ESRI 2010). I used the LS grid created in N-SPECT as input for soil loss (RUSLE) calculations in SedNet.

Vegetation cover (C) is used to solve the soil loss equation to estimate sheet-wash and rill erosion. I used N-SPECT default C values as a reference, but adjusted the values based on a national erosion study for Mexico (SEMARNAT-UACH 2002). Due to large variations in vegetation cover between different crops (SEMARNAT-UACH 2002), I calculated an average value for all cropland classes based on the proportional area covered by dominant crops within the study area.

Gully erosion

Despite the capability of SedNet to model gully erosion, spatial data on the presence or absence of gullies across the study area were not available; this is a limitation of the model application. N-SPECT does not model gully erosion.

River bank erosion

SedNet uses a map of riparian vegetation to estimate bank erosion rates (Wilkinson et al. 2004; Wilkinson et al. 2009). I created a consistent riparian vegetation map across the study area in two steps. First, I combined the polygons identified as riparian vegetation from the three land use maps. Second, I identified riparian vegetation as the portions of these polygons corresponding to tree-dominated land use classes and within an 80 m buffer along the stream network. Bank erosion is not included in N-SPECT.

Particulate nutrients

The percentage of clay and nutrient content (only nitrogen for the study) of soil units is required to estimate the amount of particulate nutrients associated with soil erosion in SedNet (Wilkinson et al. 2004). I used the Soil and Terrain Database for Latin America and the Caribbean (SOTERLAC), which provides information on some of these parameters (Dijkshoorn et al. 2005) and has been used for regional modelling with N-SPECT (Burke & Sugg 2006).

Deposition and loss of sediment and nutrients

Reservoirs are a major sink for suspended sediments transported through rivers (Walling 2006) and are used in SedNet to estimate sediment deposition based on an updated version of the Brune equation (Wilkinson et al. 2004). The surface area of reservoirs, along with mean air temperature (also from WorldClim dataset), is also used in SedNet to estimate denitrification. I mapped 11 reservoirs by combining the national reservoir database (CONAGUA 2008) and the most recent land use map (i.e., 2000), in which water bodies are delineated. Reservoir capacity (GL) was extracted from the database or estimated using ArcMap 3D Analyst (ESRI 2010) (see **Supplementary material**).

Along with reservoirs and lakes, river floodplains are landforms where substantial volumes of suspended sediments accumulate (Walling et al. 2003) and thus are incorporated into SedNet to estimate sediment deposition. Due to lack of information on the spatial distribution of floodplains, I employed the Catchment Modelling Toolkit

(CSIRO-CRC 2005) to calculate the multi-resolution valley bottom index (MrVBf) and delineate flat low-lying areas (Teng et al. 2005) which can be related to depth of soil deposit (Gallant & Dowling 2003). Based on Wilkinson et al. (2009), I used ArcMap (ESRI 2010) to identify the natural breaks (Smith 1986) in MrVBf values across the region. I selected those areas with an index value ó 2.0 as potential floodplains. Sediment deposition in floodplains is not considered in N-SPECT.

Evaluation of performance and outputs of models

Performance of models

I summarised and compared differences in performance for modelled *vs*. observed runoff and suspended sediments. I compared modelled runoff with discharge records from gauge stations (IMTA 2010). For each station I estimated the mean annual runoff (based on monthly aggregated data) and the bootstrapped 95th (percentile) confidence-interval (CI) using Pop-Toolsô (Hood 2010) Monte Carlo simulation (Manly 2007) tool (settings: data were resampled with replacement, 10,000 replicates, α = 0.05). Gauge station data were further validated against reported mean annual runoff from Mexicoôs Water Commission (CNA 2008). Of the 11 gauge stations with good data, 7 stations also included TSS records (data were limited to a few number of years of record: median of 15), which were summarised as annual sediment loads using monthly aggregates. To compare modelled and recorded sediment load estimates, I summarised annual sediment records (reported as m^3) using monthly mean volume from data recorded in the gauge stations. To calculate TSS loads (tonnes Ł year^{-1}) from gauge station data, I used the typical density of saturated sediment (1.3 tonnes Ł m^{-3}) (O'Connor et al. 2003), and estimated the mean and the bootstrapped 95th CI for the mean load following the same procedure as for runoff. I plotted observed *vs*. modelled data to assess agreement and to identify patterns of over- and under-estimation.

Spatial pattern of supply and estimated loads of pollutants

I summarised similarities and differences between models in estimates of pollutant load and overall budget. I focused on total suspended sediment (TSS) and nitrogen (dissolved inorganic nitrogen: DIN, and total nitrogen: TN) as examples of typical constituents considered in water quality modelling and monitoring (results for other modelled pollutants are provided as **Supplementary Material**). First, I compared the results based on common outputs for both models (i.e., supply of TSS from hillslope erosion and DIN from catchments and subcatchments). This identified differences in spatial distribution of supply of pollutants resulting from inherent differences between models, rather than from processes not modelled by N-SPECT (e.g., deposition, denitrification and particulate nutrients). To further explore TSS outputs of the two models, I used ArcMap Spatial Analyst (ESRI 2010) to calculate soil loss (based on the soil loss equation, RUSLE) and applied a constant sediment delivery ratio of 0.05 across the study area (i.e., the default value incorporated in SedNet) to compare this grid with the modelled supply derived from both models. Second, I compared results based on the final outputs of each model (i.e., considering both common outputs and processes modelled only in SedNet) and quantified differences in end-of-river pollutant loads, as well as the differences in modelled contributions from specific subcatchments. I used percentage contribution from subcatchments to identify the sub-catchments that were contributing most to the region-wide pollutant load. This relative measure can be used to directly compare maps from both models and to prioritise catchment management.

Results

Performance of models

Overall, SedNet consistently estimated spatial variations in runoff with higher accuracy. In most cases, SedNetôs modelled values fell within the confidence intervals for runoff calculated from gauge data (**Figure 4.3A**). I used the coefficient of efficiency (E^*) to assess the performance or goodness of fit of the regionalisation models used in SedNet (Wilkinson et al. 2006) and obtained a good fit between the runoff coefficient (Rc) and Eo/P ($E^* = 0.776$). Similarly, E^* values for bankfull flow (Q_{bf}) and median overbank

flow (Q_{mo}) (0.932 and 0.829, respectively), indicated a good fit between SedNet predictions and gauge data. E^* is similar to an R^2 and expresses the proportion of variance of the observed runoff coefficient explained by the model (Nash & Sutcliffe 1970; Wilkinson et al. 2004). In contrast, most of N-SPECTôs modelled runoff values fell outside of the confidence intervals, generally over-estimating runoff in gauges recording lower discharge and under-estimating runoff for gauge stations with higher discharge. The highest discrepancies between modelled and observed runoff (including over and under-estimation in both models) corresponded to gauges located downstream of wet areas (e.g., gauges 9067 and 9068, **Figure 4.1C**).

I found important differences in modelled and observed TSS loads for both models (**Figure 4.3B**). Estimated TSS loads derived using N-SPECT systematically fell outside the confidence intervals around observed values and above the expected range of variation, sometimes orders of magnitude above recorded data. Modelled TSS loads from SedNet were mostly within the confidence intervals, except for a couple of gauge stations (9063 and 9089). SedNet under-estimated TSS for these gauges, but I found no climatic or landscape features to explain these discrepancies. Variations in estimated TSS loads between models were in agreement with differences in modelled sediment supply described below (section 3.2). Both models exhibited greater differences between estimated and observed TSS loads than between estimated and observed runoff.

Figure 4.3. Observed and modelled flow and suspended sediment loads derived using N-SPECT and SedNet models: A) mean annual runoff depth; and B) mean total suspended sediment loads (due to large differences between observed TSS loads and loads estimated using N-SPECT, I use a logarithmic scale to visualise estimated loads). Red lines represent the bootstrapped 95th (percentile) confidence interval for the mean, estimated from gauge data (only available for records before the construction of dams).

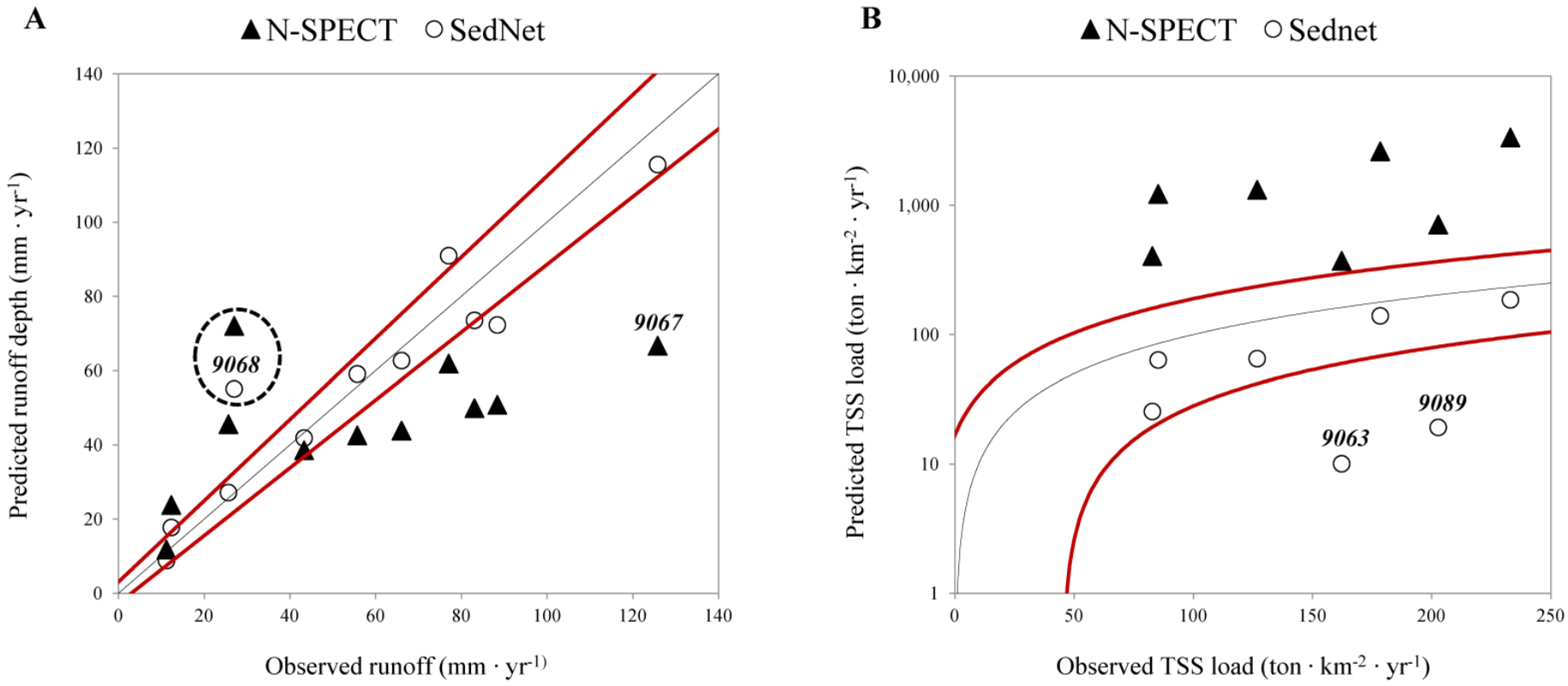

Figure 4.3. Observed and modelled flow and suspended sediment loads derived using N-SPECT and SedNet models

Spatial patterns of supply and estimated loads of pollutants

Differences in patterns of pollutant supply and contribution from subcatchments

Despite the observable spatial similarity between the two models in terms of modelled TSS supply (**Figure 4.4.4A,B**), predicted values for individual subcatchments differed markedly across the study area. Overall, the percentage difference between models was low to moderate (generally <25% difference for individual subcatchments), with N-SPECT estimating higher TSS supply per subcatchment across most of the region (**Figure 4.4.4C**). Examination of the areas where models tended to over- or under-estimate supply revealed that, in general, N-SPECT estimated lower TSS supply than SedNet from flat areas (e.g., coastal plains, identified as red and orange areas in Figure 4.4.4C). These areas ï where SedNet estimates higher TSS supply ï generally have low TSS supply. The soil loss calculated externally (i.e., applying a constant sediment delivery ratio ï SDR ï of 0.05) was consistent with the map showing TSS supply derived from SedNet (not shown). Subtracting the soil loss map calculated externally from the TSS supply map derived from N-SPECT exposed the spatially variable estimated SDR layer internally calculated by N-SPECT (**Figure 4.5A**). N-SPECTôs considerably higher SDR (median = 0.89) applied across the landscape (**Figure 4.5B**) explains its over-estimation of TSS loads (**Figure 4.3B**), as well as the observed lower contributions by subcatchments in flat areas in relation to SedNet.

Figure 4.4. Patterns across subcatchments of supply of hillslope suspended sediment (TSS) and dissolved inorganic nitrogen (DIN), as modelled by N-SPECT and SedNet. In all cases supply is represented as the percentage contribution by each subcatchment to the total TSS or DIN supply across the study area: A) hillslope TSS supply estimated by N-SPECT; B) hillslope TSS supply estimated by SedNet; C) difference in percentage TSS supply per subcatchment (SedNet - N-SPECT); D) DIN supply estimated by N-SPECT; E) DIN supply estimated by SedNet; and F) difference in percentage DIN supply per subcatchment (SedNet - N-SPECT).

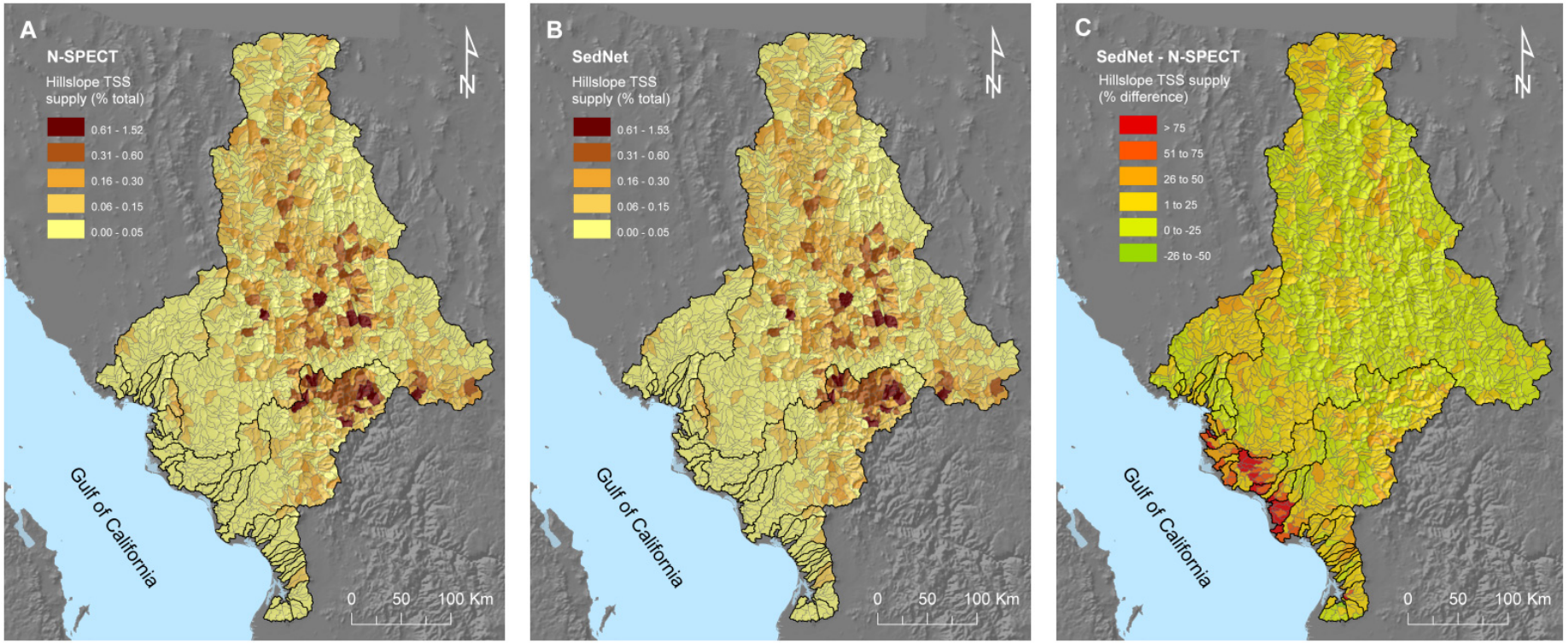

Figure 4.4. Patterns across subcatchments of supply of hillslope suspended sediment (TSS) and dissolved inorganic nitrogen (DIN)

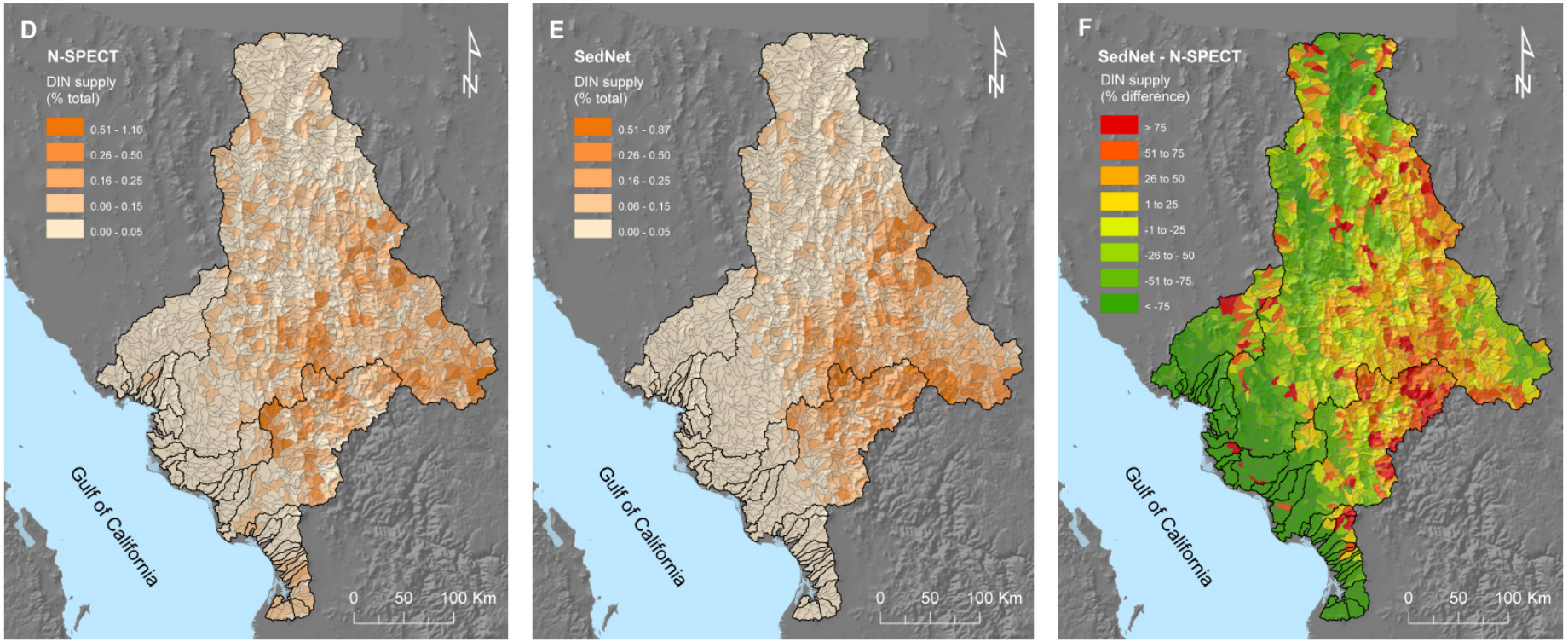

Figure 4.4. Patterns across subcatchments of supply of hillslope TSS and DIN, as modelled by N-SPECT and SedNet (cont.)

Although spatial patterns in estimated DIN supply were broadly similar between the two models (**Figure 4.4D, E**), differences for individual subcatchments were more pronounced and widespread than for TSS (**Figure 4.4F**). DIN supply as calculated by SedNet was concentrated strongly in eastern mountainous and wet subcatchments (**Figure 4.4E**), while N-SPECT modelled a somewhat less concentrated pattern of supply (**Figure 4.4D**). The more dispersed pattern from N-SPECT included larger contributions from some western and south-eastern catchments, in most cases clearly driven by the dominant land uses (e.g., agriculture and pasture; see **Figure 4.1C**). Neither model systematically estimated higher or lower DIN supply across the domain.

Figure 4.5. Sediment delivery ratios (SDRs) from N-SPECT and differences in SDR between models. A) spatially variable SDR automatically calculated by N-SPECT; B) absolute difference between SedNet default SDR (0.05) and N-SPECT estimated value. Yellow denotes subcatchments where SedNet SDR is higher (a negligible area). Areas in different tones of green indicate larger SDR values for N-SPECT (where most of the domain has SDR values >0.6 as estimated by N-SPECT).

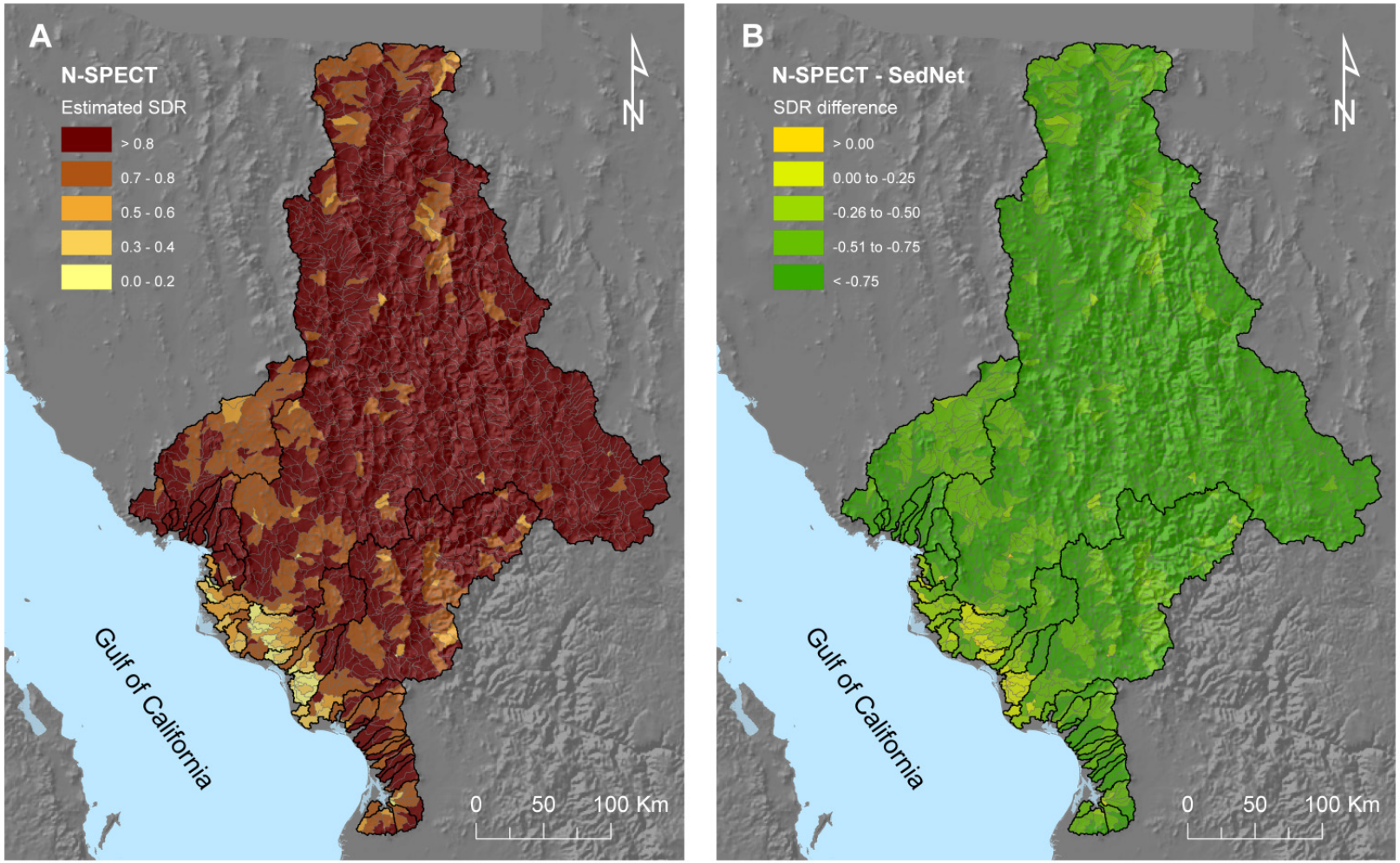

Figure 4.5. Sediment delivery ratios (SDRs) from N-SPECT and differences in SDR between models

Comparing modelled TSS and total nitrogen (which incorporates dissolved inorganic nitrogen, dissolved organic nitrogen, and particulate nitrogen) between models revealed the strong influence of reservoirs and the contribution of particulate nutrients (associated with erosion) to the total nutrient load, both of which are incorporated only in SedNet (**Figure 4.6**). The influence of large reservoirs was evident from the low TSS contribution estimated by SedNet for most of the Yaquiôs upper catchment (green areas in Figure 4.6C) and to a lesser extent for the upper Mayo River catchment (yellow in south-east part of Figure 4.6C). Accounting for reservoirs in SedNet shifted the distribution of areas of high TSS contribution from the upper to the lower portions of the catchments compared to N-SPECT (red areas in **Figure 4.6C**). The contribution of river bank erosion estimated in SedNet was negligible (< 1% of TSS load), explaining the minimal differences in patterns of TSS supply between the summed (bank and hillslope) and the hillslope-only erosion supply (maps not presented).

Figure 4.6. Patterns of supply and contribution by subcatchments of total suspended sediment (TSS) and total nitrogen (TN), as modelled by N-SPECT and SedNet, with SedNet incorporating sinks and additional pollutant sources. For SedNet, the term ñcontributionò is used instead of ñsupplyò to indicate that SedNet considers sinks (e.g., sediment deposition and denitrification) that modify subcatchmentsô supply in terms of their contributions to end-of-river amounts. N-SPECT does not consider sinks, so supply equals contribution. Supply is represented as the percentage contribution by each subcatchment to the total TSS or DIN supply across the study area: A) TSS supply estimated by N-SPECT; B) TSS contribution estimated by SedNet, including sediment eroded from river banks and deposition in reservoirs and floodplains; C) difference in percentage TSS contribution (SedNet - N-SPECT); D) TN supply estimated by N-SPECT (calculated using Event Mean Concentration for TN); E) TN contribution estimated by SedNet as the sum of dissolved inorganic nitrogen (DIN), dissolved organic nitrogen (DON) and particulate nitrogen (PN), considering sinks (i.e., DIN denitrification, DON loss in reservoirs and PN deposition in reservoirs and floodplains); and F) difference in percentage TN contribution (SedNet - N-SPECT).

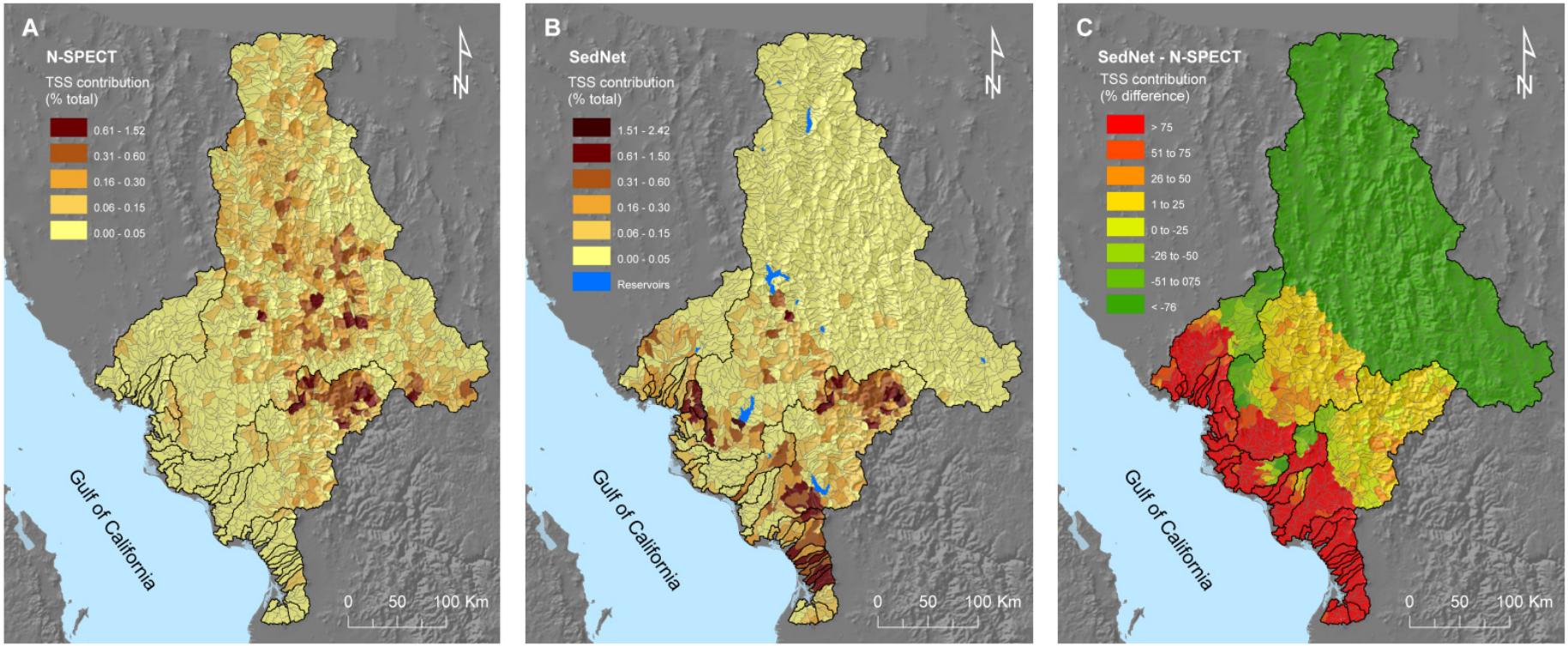

Figure 4.6. Patterns of supply and contribution by subcatchments of total suspended sediment (TSS) and total nitrogen (TN)

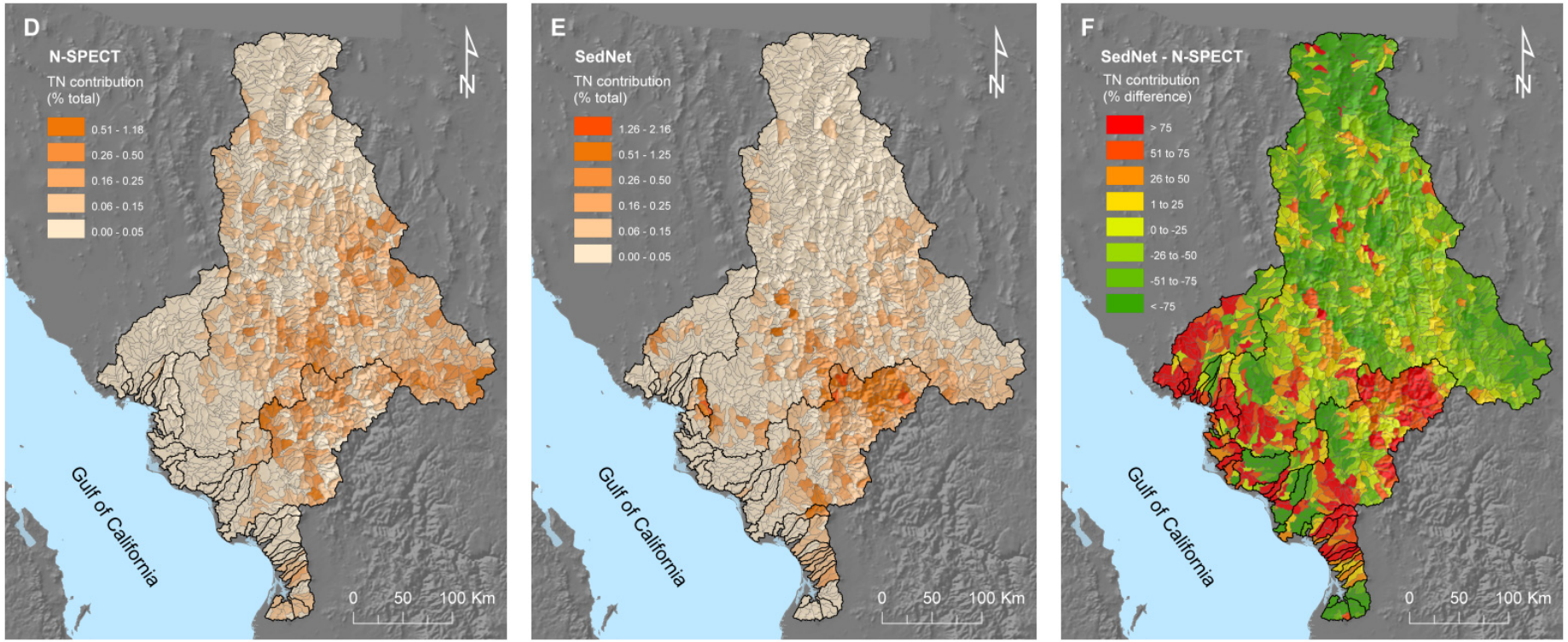

Figure 4.6. Patterns of supply and contribution by subcatchments of TSS and TN (cont.)

Differences in estimated end-of river loads

Differences in the total (end-of-river) loads for both pollutants were considerable (**Figure 4.7**), highlighting the more conservative estimates from SedNet of TSS supply to the stream network. In addition to the large differences in estimated TSS loads (up to 3 orders of magnitude), the relative contributions from catchments also varied considerably (**Figure 4.7A,B**). Major differences were notable in loads of the three major catchments, where the largest catchment (Yaqui) was identified as the major contributor to end-of-river loads in N-SPECT, while SedNet estimated that most of the sediments effectively delivered to the coast were from the second largest catchment (Mayo). This difference was mostly driven by deposition occurring in reservoirs. Differences in overall patterns across catchments of estimated total nitrogen loads were less marked, with the two largest catchments identified by both models as the main contributors (**Figure 4.7C,D**). However, two important differences are noteworthy: first, the contributions from these two very large catchments in SedNet was almost identical (~1,000 ton/year), while N-SPECT estimated total nitrogen load from the Yaqui as three times larger than from the Mayo. Differences in total nitrogen from the smaller coastal catchments were less pronounced.

Figure 4.7. End-of-river loads of total suspended sediment (TSS) and total nitrogen (TN) by catchment. A) TSS loads estimated by N-SPECT; B) TSS load estimated by SedNet including sediment eroded from river bank and deposition in reservoirs and floodplains; C) TN load estimated by N-SPECT (calculated using Event Mean Concentration for TN); and D) TN load estimated by SedNet (DIN + DON + PN - DIN denitrification - DON reservoir loss - PN deposition in reservoirs and floodplains).

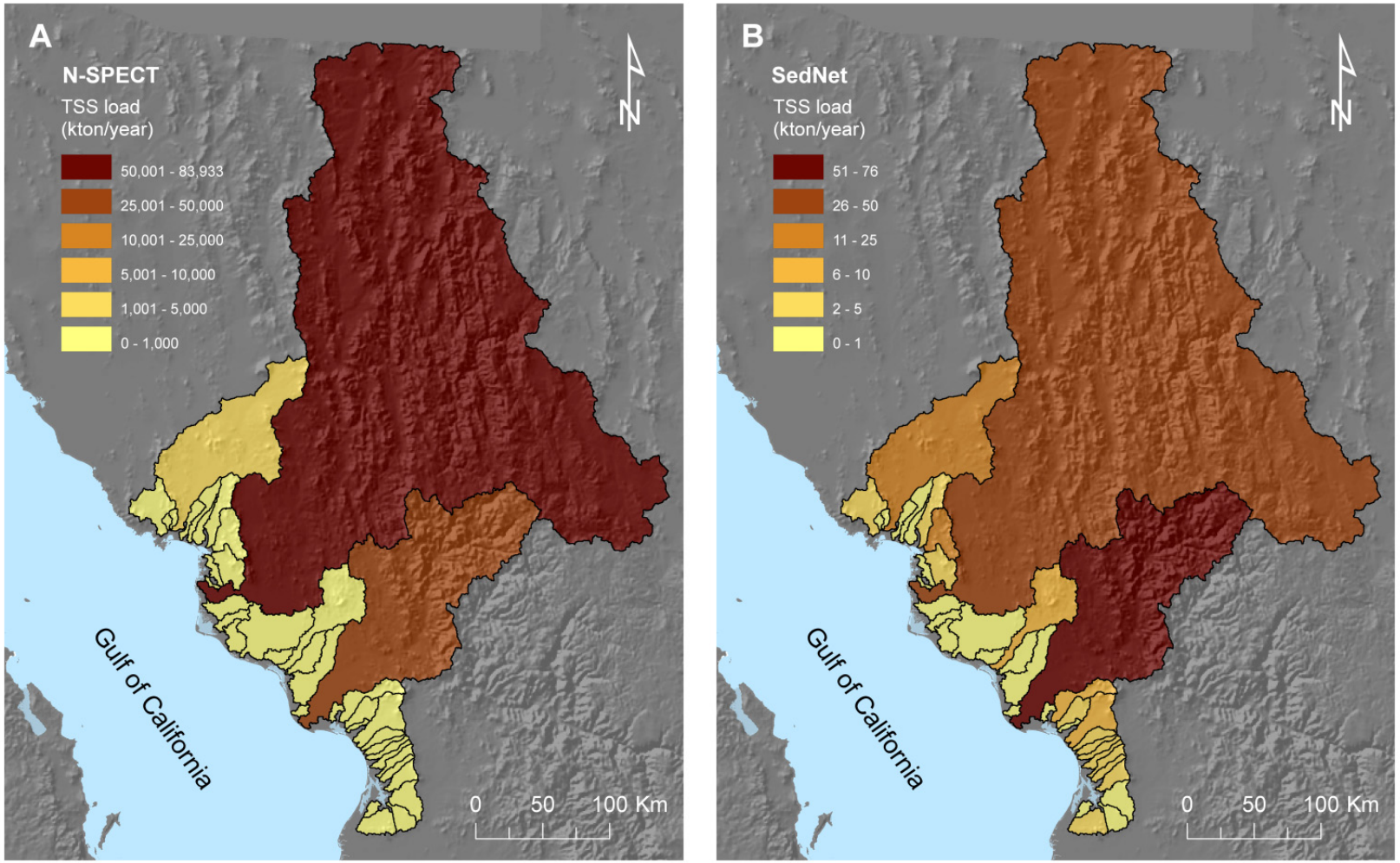

Figure 4.7. End-of-river loads of total suspended sediment (TSS) and total nitrogen (TN) by catchment

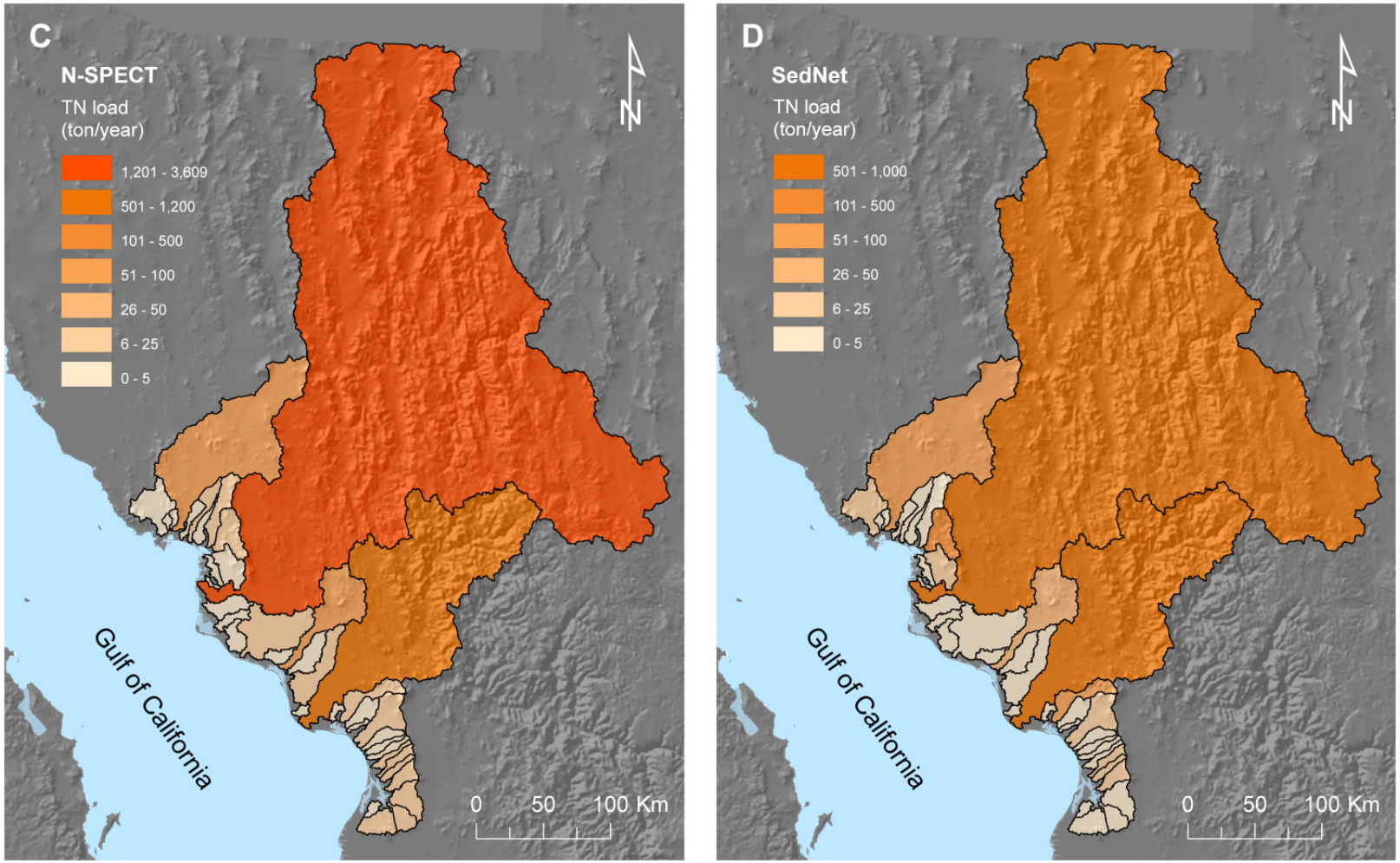

Figure 4.7. End-of-river loads of total suspended sediment (TSS) and total nitrogen (TN) by catchment (cont.)

Discussion

In areas where land-based impacts on coastal-marine ecosystems are important (Halpern et al. 2009), a crucial yet neglected component of conservation planning is to reduce or mitigate such impacts (Alvarez-Romero et al. 2011). I compared two catchment models to assess their performance, data requirements, ease of implementation, and utility for regional-scale planning, and I found important differences. Here I discuss the implications of the key findings for conservation practitioners seeking to incorporate land-based impacts into coastal-marine exercises in conservation prioritisation.

Differences in model performance (accuracy and spatial patterns)

I found important differences in performance between models. Overall, SedNet performed better than N-SPECT. SedNet consistently estimated spatial variations in runoff with higher accuracy and, unlike N-SPECT, its modelled TSS loads were mostly within the expected range of variation. Differences in performance between models relate to intrinsic differences in their approaches. For example, SedNet was better able to incorporate runoff processes operating at catchment scales and over longer periods of time (decades) (Wilkinson et al. 2006), and, unlike N-SPECT, SedNet can incorporate sinks (reservoirs and floodplains) and other sources of pollutants (river bank erosion and particulate nutrients). These additional considerations in SedNet reflects the ôbudgetô approach of the model, which confers a better understanding of the dominant sources, sinks and fluxes of material throughout the catchments (Walling et al. 2011). In contrast, factors negatively affecting the performance of N-SPECT in predicting long-term runoff over large areas included coarse and poor-quality spatial datasets and un-calibrated parameters. These differences further emphasise the need to pay special attention to calibrating runoff to obtain more reliable estimates of pollutant supply and delivery.

Data requirements and ease of implementation

Overall, data requirements were not considered an important limitation to the implementation of either N-SPECT or SedNet in the study area, where relatively limited data were available compared to other regions where the models have been used (Carter et al. 2005; McKergow et al. 2005b). In most cases, datasets of appropriate resolution for region-wide applications were available or it was possible to model the required data (e.g., locations of floodplains and riparian vegetation) with readily available tools and data. However, these models of required inputs should be considered proxies and require validation for subsequent work. Some datasets were not available (e.g., gullies) and likely limit the accuracy of model outputs (Caitcheon et al. 2012; Wilkinson et al. 2009). Once the datasets were compiled, running the models was relatively straightforward. However, data requirements and parameterisation demanded more time and expertise for SedNet than N-SPECT.

The time commitment to locate, assemble, assess and model data should not be underestimated. Catchment modelling is a major undertaking that requires not only substantial time, but also technical expertise in geographic information systems and modelling. Assembling a research or planning group with expertise in areas relevant to this task (e.g., hydrology, water quality, soil science) can greatly facilitate data preparation and improve modelling outputs. Also, exploration of intermediate model outputs and comparison of modelling outputs with the results of other studies using the same models can help to identify errors in data inputs or parameterisation and explain unexpected outputs (e.g., very high values). While time consuming, the process of compiling and preparing data can be important to identify gaps to guide further studies in the region. Despite these challenges, coastal-marine conservation planners recognise the importance of such modelling endeavours (Halpern et al. 2009; Tallis et al. 2008).

Applicability of outputs to planning

Outputs from both models were appropriate to represent broad spatial patterns of pollutant supply across the study area, but spatial differences in model outputs have implications for marine conservation planning. For example, spatial patterns of TSS from subcatchments varied between models, with the outputs of SedNet indicating that TSS loads were highest next to the regionôs highest marine conservation priority areas.

Consequently, these catchments would hopefully be prioritised for management of water quality to minimise land-based impacts on these high-priority marine regions. However, if N-SPECTôs results were used instead, these catchments would not be prioritised. Assuming N-SPECTôs outputs are less reliable than those of SedNet, which seems likely from my comparisons, prioritisation of catchment management based on N-SPECT would lead to high-priority marine regions continuing to suffer excessive loads of sediment and nutrients. Alternatively, N-SPECTôs predictions could lead to these marine priority areas being excluded as potential marine conservation areas if land-based pollution was seen to be intractable. Thus reliable estimates of nutrient and sediment inputs, and end-of-river loads in particular, are very important for marine conservation planning applications so that affected areas can be linked to the sources of pollutants (Alvarez-Romero et al. 2012). Ideally load data can be incorporated into flood plume models to more fully assess vulnerability of marine areas to land-based threats (Burke & Sugg 2006; Devlin et al. 2012b; Maughan & Brodie 2009) and to more comprehensively link affected marine areas to pollution sources.

Conclusions

My study was focused on integrating land-based impacts into coastal-marine conservation planning. The next steps towards truly integrated land-sea planning involve linking land-based impacts to marine regions (i.e., through river plume models: Devlin et al. 2012b), assessing where the impacts negatively affect sensitive and vulnerable coastal and marine habitats or species (Halpern et al. 2007), and prioritising catchments for management to reduce land-based impacts (Klein et al. 2010). Different strategies to address cross-system threats are possible, depending on factors such as irreplaceability of the affected marine features and the feasibility to mitigate land-based and/or marine threats (Alvarez-Romero et al. 2011). The configuration of priority maps associated with these strategies can be very different and spatially uncorrelated, hence there will be tradeoffs associated with the decisions being made. The exploration of spatial options under different scenarios will provide critical information to guide the planning process. No study that I am aware of has included all of these steps, thus exploring options to integrate these elements in coastal and marine planning should be considered a research priority.

Chapter 5. A combined remote sensing-GIS approach to model exposure of marine ecosystems to riverine flood plumes

Abstract

Increased loads of land-based pollutants are a major threat to coastal-marine ecosystems. Identifying the affected marine areas and the scale of influence on marine ecosystems is critical to assess the ecological impacts of degraded water quality and to inform planning for catchment management and marine conservation. Studies using remotely sensed data have contributed to our understanding of the occurrence and influence of river plumes, and to our ability to assess exposure of marine ecosystems to land-based pollutants. However, refinement of plume modelling techniques is required to improve risk assessments. I developed a novel approach to model exposure of coastal-marine ecosystems to land-based pollutants. I used supervised classification of MODIS-Aqua true-colour satellite imagery to map the extent of plumes and to qualitatively assess the dispersal of pollutants in plumes. I used the Great Barrier Reef (GBR), the world's largest coral reef system, to test our approach. I combined frequency of plume occurrence with spatially distributed loads (based on a cost-distance function) to create maps of exposure to suspended sediment and dissolved inorganic nitrogen. I then compared annual exposure maps (2007-2011) to assess inter-annual variability in the exposure of coral reefs and seagrass beds to these pollutants. My findings indicate that classification of true-colour satellite images is useful to map plumes and to qualitatively assess exposure to land-based pollutants. This approach should be considered complementary to remote sensing methods based on ocean colour products used to characterize surface water in plumes. Observed inter-annual variation in exposure of ecosystems to pollutants stresses the need to incorporate this temporal component into plume exposure/risk models. My study contributes to our understanding of plume spatial-temporal dynamics of the GBR and offers a method that can improve plume exposure models. The proposed method can also be applied to monitor exposure of coastal-marine ecosystems to plumes and explore their ecological influences.

Introduction

Land-based pollution poses a major threat to marine ecosystems globally and is projected to increase with the expansion and intensification of agriculture, grazing, and urbanization (Butchart et al. 2010; CBD 2006; Halpern et al. 2008; MEA 2005). Increasing loads of land-based pollutants (e.g., nutrients, sediment, pesticides) associated with land use change are causing major ecological impacts, potentially interacting with other stressors (e.g., overfishing, increased ocean temperature, ocean acidification, sea level rise). Improving coastal-marine water quality is therefore a global priority (UNEP 2012). Nutrient runoff from intensive use of fertilizers in agriculture can cause severe eutrophication, toxic phytoplankton blooms, and hypoxia in coastal and marine ecosystems (Cloern 1996; Diaz & Rosenberg 2008; Gabric & Bell 1993; Howarth 2008). Nutrient enrichment of coastal waters has also been related to increased risk of coral bleaching (Wooldridge 2009) and outbreaks of the reef-destroying crown-of-thorns starfish (COTS) (Brodie et al. 2005; Fabricius et al. 2010). Globally, land use change associated with agriculture, forestry, and urbanization have led to significant soil loss, resulting in increased sediment loads and turbidity of coastal waters (Croke & Hairsine 2006; Thrush et al. 2004; Walling 2006), which in turn have been related to reduced coral recruitment (Hutchings et al. 2005), increased seagrass mortality (Cabaco et al. 2008), and declines in invertebrate densities (Reid et al. 2011). Pesticides are a major concern, particularly for sites adjacent to coastal catchments with intensive agriculture or forestry plantations, and have been related to dieback in mangroves, as well as reduced photosynthesis and expulsion of the symbiotic algae (bleaching) from corals (Duke et al. 2005; Haynes et al. 2007; Hutchings et al. 2005).

Dispersal of land-based pollutants has been traced to offshore waters, sometimes hundreds of kilometers from river mouths (Beman et al. 2005; Brodie et al. 2010; Devlin & Schaffelke 2009), highlighting the need to consider such threats in marine conservation planning. The ecological impacts of land-based pollutants vary not only with the magnitude and extent of riverine influence, but also with the coastal-marine ecosystems being affected (Halpern et al. 2007; Halpern et al. 2008), and the frequency and duration of plume occurrence (McKenzie & Unsworth 2011). Thus considering land-based threats is of utmost importance when planning for coastal-marine

conservation and catchment management (Alvarez-Romero et al. 2011). Ignoring the potential impacts of land-based threats can reduce the effectiveness of marine conservation areas (Stoms et al. 2005; Wilson et al. 2005a), especially when these threats are extensive (Halpern et al. 2009). The consequences of ignoring these threats, and thus of the ecological changes associated with decreased water quality, reduce the effectiveness of marine conservation areas (Brodie et al. 2012b). Land-based pollution not only impacts marine biodiversity but also affects human livelihoods by compromising marine ecosystem services, such as food provisioning, recreation, and protection against coastal hazards (Lubchenco et al. 2003; Worm et al. 2006).

Identifying marine areas and ecosystems affected by land-based impacts is crucial if management is to address those impacts. Mapping river plumes ï their extent and surface water characteristics ï is a key approach to identify the flow of land-based pollutants into marine systems (Devlin et al. 2011). Several approaches have been developed to map river plumes, but improvements are possible. A common approach has been the manual delineation of plumes based on aerial photography and satellite imagery (Devlin & Schaffelke 2009; Devlin et al. 2011). A key limitation of visual interpretation (i.e., manual delineation) of plumes is that it requires considerable experience and leads to human error; it is also time-consuming, thus limiting the number of images that can be processed. Some notable advances have been made in automating the plume identification process through the use of ocean-color products derived from satellite images (e.g., estimated chlorophyll and suspended sediment), and has been successfully used to map the freshwater extent of plumes (Schroeder et al. 2012) and to monitor variations in surface water parameters (Brando et al. 2010). However, to date these methods are limited because estimated parameters (e.g., chlorophyll, suspended sediment) can be associated with multiple processes (not only related to plumes) and, in general, perform poorly under certain atmospheric and water (e.g., moderate cloud cover and sun glint, high turbidity characteristic of the initial phases of plumes) conditions. Hydrodynamic models have also been used (in combination with estimated pollutant loads discharged by rivers) to estimate the dispersal of pollutants and hence to assess exposure of marine ecosystems to land-based pollution (e.g., Cherubin et al. 2008). However, this approach generally requires significant expertise and computational resources and, in most cases, has not been validated against in situ data due to time and resources required for regional studies.

Thus I explore a novel approach of automating river plume mapping that can overcome some of these limitations.

The purpose of my study is to develop a novel method to assess exposure of coastal-marine ecosystems to land-based pollutants. This approach, based on the classification of true-colour satellite imagery, offers a way of mapping plumes under moderate cloud cover and with sun glint, as well as in highly turbid waters where the performance of other remote sensing methods is limited. More specifically, I aim to improve existing plume mapping and estimating exposure to river-borne pollutants in two ways: (1) by developing a relatively simple method to map the extent of land-based pollutants and qualitatively assess their dispersal; and (2) by assessing spatial-temporal variability in the exposure of marine ecosystems (coral reefs and seagrass beds) to plumes, key to assessing potential impacts. My study contributes to our understanding of the spatial-temporal dynamics of plumes in the Great Barrier Reef (GBR), Australia. I discuss the application of this method to monitor exposure of marine ecosystems to plumes and to study ecological responses to land-based pollutants.

Methods

Study region

My study focuses on the GBR, the world's largest coral reef system, located in the Coral Sea, off the coast of Queensland in north-east Australia, and a World Heritage Site since 1981. The GBR is globally recognized for its outstanding ecological, cultural and socioeconomic values (GBRMPA 2009) and since 1975 a large part of the reef is protected by the Great Barrier Reef Marine Park (GBRMP). The GBRMP includes around 2,900 reefs and 940 islands and cays, covering almost 345,000 km2. The sheer extent of the park helps to limit the impact of human use, such as fishing and tourism (Day 2011). The rezoning of the GBRMP vastly increased the area under strict protection (i.e., no-take areas) from 4.5% to >33% (Fernandes et al. 2005), with documented benefits for marine biodiversity and fisheries (McCook et al. 2010). However, threats originating from land-based activities (e.g., coastal development and decreased water quality associated with agriculture and grazing in coastal catchments),

and from global climate change (e.g., increased water temperature and ocean acidification) are major concerns (Brodie et al. 2012b; GBRMPA 2009). Since the arrival of Europeans in Australia, loads of suspended sediments delivered to the GBR have increased drastically following extensive land clearing for grazing, agriculture and coastal development (Joo et al. 2012; Kroon et al. 2012). Similarly, fertilized cropping has significantly increased the loads of nutrients (particularly dissolved inorganic nitrogen and phosphorus), thus resulting in eutrophication of estuarine and marine environments (Brodie et al. 2011; Brodie et al. 2012b). Pesticides washed from agricultural lands add to the load of land-based pollutants delivered into the sea (Kennedy et al. 2012; Lewis et al. 2009). Most of the pollutants are delivered during the wet season and in particular during peak-flow periods, when large plumes can be identified along the coast (Devlin & Brodie 2005), occasionally reaching offshore reefs (Devlin et al. 2012b; Schroeder et al. 2012).

I focus my 5-year study (2007-2011) on four of the six Natural Resource Management regions (hereafter NRMs) of the GBR: Wet Tropics, Burdekin, Mackay-Whitsundays, and Fitzroy (**Figure 5.1**). The Cape York and Mary-Burnett NRMs were excluded from the study because information on estimated annual loads was not available or incomplete. The four selected NRMs illustrate a range of land use and landscape characteristics (e.g., climate, soils, topography) that result in different patterns of delivery of pollutants by major rivers across the GBR (Brodie et al. 2009; Kroon et al. 2012). Differences in flow are reflected in the spatial and temporal distribution and composition of river plumes observed along the GBR coast (Devlin et al. 2011; Devlin et al. 2001). Along this wide latitudinal range (10Á41ô20ò S to 24Á30ô00ò S) are also differences in the distribution of marine ecosystems (e.g., coral reefs in the north are closer to the coast), which means that some of these ecosystems are more likely to be affected by plumes of comparable magnitude. These characteristics make the GBR an ideal area to study plume dynamics across large areas and to illustrate the application of the plume exposure model for regional planning.

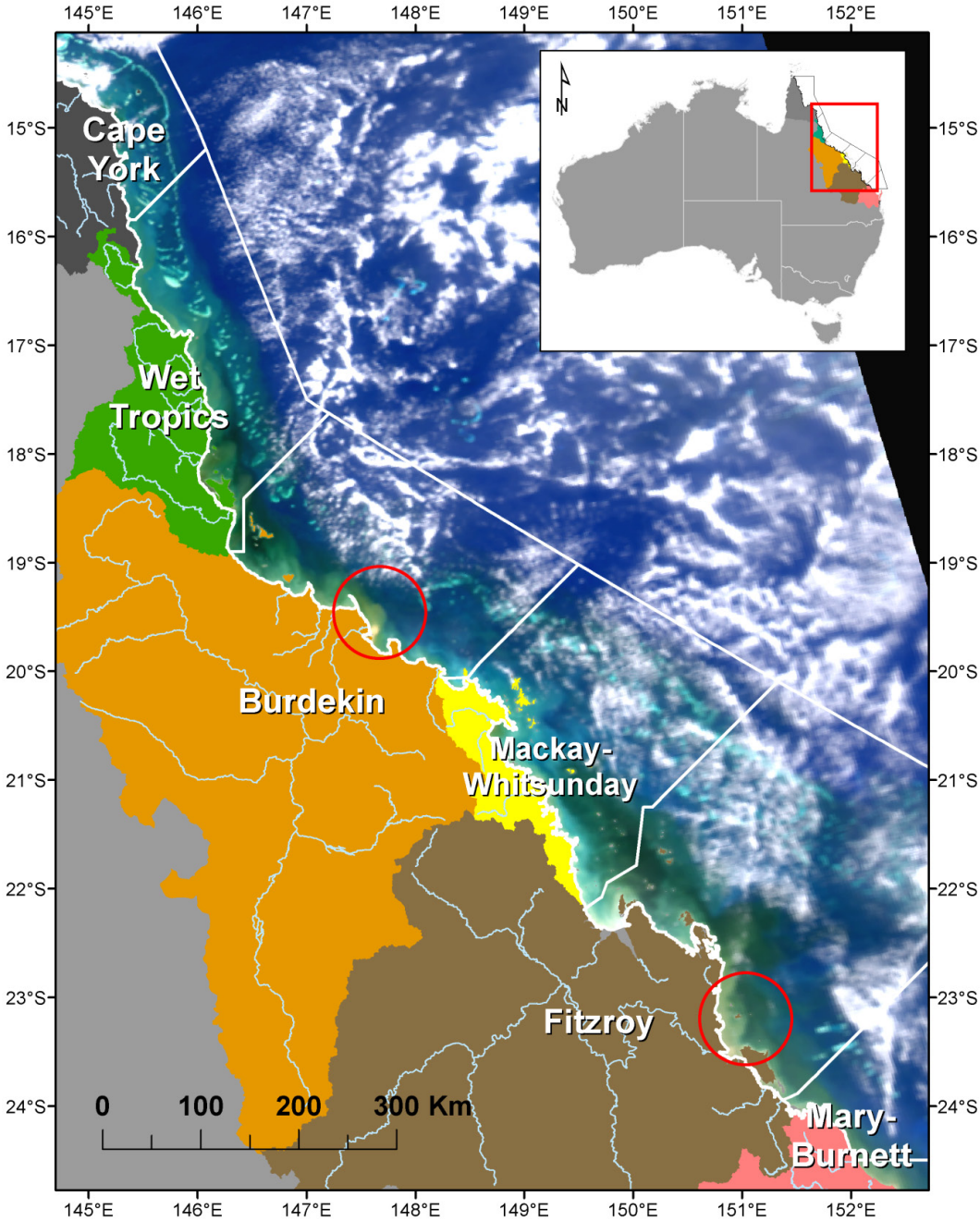

Figure 5.1. MODIS-Aqua satellite image (from NASA OceanColor online catalogue) illustrating river flood plumes along the coast of the Great Barrier Reef Marine Park, Australia on January 25th, 2011. The presence of high suspended sediment areas ï identifiable as brown areas (red circles) close to river mouths ï is more evident for the two largest rivers, the Burdekin and Fitzroy. The six coastal NRMs of the GBR catchment are shown in different colours. These NRM regions extend seaward and are limited by the GBR Marine Park boundary (delineated by white lines) thus providing management areas which are aligned with catchment practices.

Modelling exposure to plumes

My exposure model builds on previous models developed for the GBR (i.e., Devlin et al. 2012b; Maughan & Brodie 2009), but includes several improvements. I selected two pollutants for in-depth examination because of their known impacts on the GBR: total suspended sediment (TSS) and dissolved inorganic nitrogen (DIN) (Brodie et al. 2012b; Fabricius 2011). TSS and DIN are commonly measured parameters in water quality monitoring programs. In the GBR, loads of these parameters vary significantly between marine areas adjacent to NRMs in the GBR (Joo et al. 2012; Waterhouse et al. 2012). These parameters also illustrate different dispersal patterns of pollutants and hence spatial patterns of exposure across the GBR (Devlin et al. 2001; Devlin & Brodie 2005). My analysis focused on the wet season (December to April inclusive) and, unlike previous studies limited to low cloud-cover days (Devlin et al. 2012b; Maughan & Brodie 2009), used all available MODIS-Aqua satellite images over the GBR (usually 2, but up to 4 images/day). Restricting the analysis to summer months minimised problems associated with sunlight reflection on the ocean surface (i.e., the solar zenith angle is smallest and albedo problem is limited). It also minimised the occurrence of ñfalseò plume areas associated with wind-driven re-suspension of sediments during the dry season (Larcombe et al. 1995; Orpin et al. 1999). Almost 2,100 images were available for the GBR region for the study years (2007-2011), but not all images contained data within the boundaries of the study area. The main output of my model is a qualitative map indicative of annual exposure of any given area within the GBR to TSS and DIN. The model is constructed in five steps (**Figure 5.2**), each of which is described in the sections that follow.

Figure 5.2. Summary of the process followed to build the plume exposure model. Different colours of boxes represent: steps (light grey), key analyses within steps (white), intermediate outputs (dark grey), and final output (black).

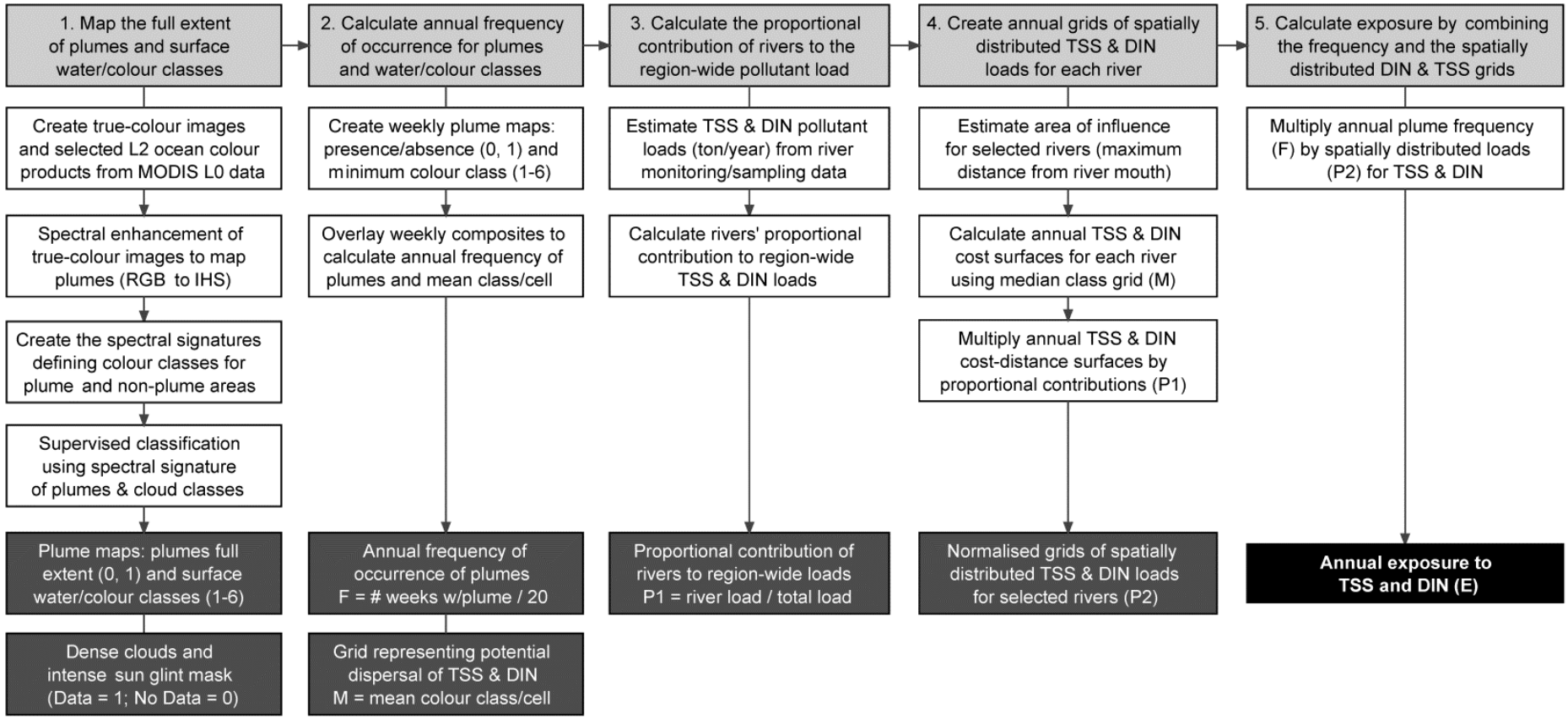

Figure 5.2. Summary of the process followed to build the plume exposure model

Map the full extent of plumes and surface water/colour classes

The first step to model exposure is to identify (map) existing plumes, as well as clouds and sun glint, in each of the ~2,100 satellite images. This step also includes mapping the different surface water masses observable within plumes (represented by colour classes), which are later used to represent the dispersal of pollutants. I used freely available satellite imagery for my analysis, and mapped the extent of river plumes with four analyses.

First, I created the images necessary for subsequent analyses. I used raw data (Level-0, hereafter L0) from the Moderate Resolution Imaging Spectroradiometer (MODIS) on-board NASAôs Earth Observation System Aqua spacecraft (oceancolour.gsfc.nasa.gov/cgi/browse.pl) to derive two types of images necessary for my model: (1) quasi-true colour (hereafter true-colour) images (~390m resolution); and (2) ocean colour products (Level-2, hereafter L2 parameters), derived using the SeaWiFS Data Analysis System (SeaDAS v6.2, Baith et al. 2001). Using the latter, I produced two L2 parameters (**Table 5.5.1**) after correcting for atmospheric contaminations. The correction involved the combined near infrared to short wave infrared atmospheric correction scheme adapted to both clear and turbid waters, commonly found in the nearshore regions of the GBR (Wang & Shi 2007). I used a modified script of the Marine Geospatial Ecology Tools (MGET, Roberts et al. 2010) to convert the swath-formatted L2 parameters into points, which I then used in ArcMap (ESRI 2010) to explore the relationship between the L2 parameters and the mapped colour classes (**Supplementary Material**).

Table 5.1. Selected L2 parameters derived from MODIS satellite imagery. The gradient in L2 parameters is considered to be indicative of variations in water quality within plumes (Devlin et al. 2012b), and thus was used to identify the plume boundary and to represent the potential dispersal of pollutants.

L2 parameter	Description	Application in plume mapping	Algorithms
nLw_667	Normalised water-leaving radiance at 667 nm	Normalised water-leaving radiance at higher wavelengths (555 nm or 667 nm) is sensitive to suspended sediment in the water column and is correlated with plume location (Salisbury et al. 2004; Thomas & Weatherbee 2006). I use this parameter as a proxy for TSS and to approximate the dispersal of sediment within the plume.	Defined as upwelling radiance just above the sea surface, in the absence of an atmosphere, and with the sun directly overhead (Gordon & Wang 1994).
adg_443_qaa	Coloured dissolved organic and detrital matter (CDOM +D) absorption coefficient at 443 nm	CDOM has been proposed as a proxy for salinity and thus been used to estimate the maximum extent of river plume influence (Schroeder et al. 2012). I use it to estimate the dispersal of dissolved inorganic nitrogen assuming conservative mixing.	Derived using the quasi-analytical algorithm (QAA) model (Lee et al. 2002), which estimates absorption due to gelbstof and detrital material at sensor wavelength of 443nm.

Table 5.1. Selected L2 parameters derived from MODIS satellite imagery

Second, I prepared the true-colour images for plume mapping. I spectrally enhanced the true-colour images using Image Analysis for ArcGIS (ERDAS 2010), which involved converting images from the Red-Blue-Green (RGB) to the Intensity-Hue-Saturation (IHS) colour space. Images converted from RGB to IHS have been used to identify and track the movement of river plume fronts (Morichon et al. 2008). The same true-colour images were used to map clouds and sun glint, but these were not enhanced for this purpose. Using RGB instead of IHS images was more conservative because larger areas get identified as clouds/sun glint, which are in turn masked out to minimise classification errors.

Third, I defined colour classes corresponding to plume and non-plume areas, as well as to dense clouds and sun glint, and created a spectral signature for each class. I did this using an image with large plumes occurring along the whole GBR coast (**Figure 5.1**) to ensure that colour variations within plumes along the latitudinal gradient were incorporated into the spectral signature. The selected image included large areas with no plumes, varied atmospheric conditions (light to dense clouds, haze and sun glint), and sections with no data (not covered by satellite swath). To create the spectral signatures, I used the ArcMap Spatial Analyst (ESRI 2010) isodata clustering tool to perform an unsupervised classification of the selected image. Unsupervised classification tries to find structure to unlabelled data (in my case to identify colour classes for plumes and non-plume areas) through an iterative optimisation clustering procedure (Ball & Hall 1965). The resulting structure then allows characterisation of the natural groupings of cells (i.e., pixels within an image) in multidimensional attribute space, in this case the IHS and RGB spaces, for plumes and clouds/sun glint, respectively.

I assessed plume maps produced using different number of classes based on two criteria: how well the mapped classes identified the river plume boundary (I assessed the classified images against visually interpreted true-colour imagery); and whether the variation of selected L2 parameters among the colour classes showed the expected gradient (as described by Devlin et al. 2012b; see section 3.1.1). For each classification I plotted the mean value of the two L2 parameters for each colour class. Due to reflectance similarities between land and very turbid plumes occurring in the mouth of the rivers, I classified the full image, without masking out land. This allowed us to map very turbid/high TSS plume areas commonly found near river mouths, which are

regularly flagged as land or very dense clouds (e.g., Devlin et al. 2012b; Okada et al. 2003; Schroeder et al. 2012). I selected the classification based on 6 classes as the most appropriate for plume mapping.

Finally, I used supervised classification to map the full extent of plumes, as well as to create a mask representing dense clouds and intense sun glint. Supervised classification uses labelled training data (i.e., the colour classes defined in the previous step) to create a spectral signature for each class, which is then used to classify all of the input imagery (~2,100 images) into colour classes (i.e., 6-class plumes, non-plume areas, and clouds/sun glint). I used the ArcMap Spatial Analyst maximum-likelihood classification tool (ESRI 2010) to produce: (1) daily 6-class plume maps representing variations in L2 parameters (also used to identify the plume boundary) and (2) masks representing dense clouds and intense sun glint, used to eliminate areas with insufficient information to map plumes. I selected a number of images covering different years, regions and months to confirm the plume extent and overall congruence of my classified plume maps against plume maps produced using the method proposed by Devlin *et al*. (2012b), as well as to visually validate the clouds/sun glint masks.

Calculate annual frequency of occurrence for plumes and colour classes

In the second step of the process (**Figure 5.2**) I used the mapped plumes to assess (on a cell-by-cell basis) how frequently they occurred, as well as to identify the most commonly observed colour classes for each year. To do this, I created weekly composite images (using the daily 6-class plume maps) to minimise the amount of area without data per image due to masking of dense cloud cover, common during the wet season (Brodie et al. 2010), and intense sun glint. I used the minimum colour class value of each cell/week to ensure mapping the colour class that represents the highest level of exposure to pollutants observed at any day during the corresponding week (i.e., assuming the colour gradient represents a gradient in exposure to pollutants). I then overlaid weekly composites (i.e., presence/absence and minimum class value) to calculate: annual frequency of occurrence of plumes (F), representing the proportion of weeks plumes were present in each cell during the wet season; and annual mean value of colour class (M), which was used to create a grid (i.e., a map in a raster data storage

format, where each cell or pixel of the map is equal sized and has a numeric value) representing potential dispersal of pollutants (section 2.2.4).

Calculate the proportional contribution of rivers to the region-wide TSS/DIN load

In my third step (**Figure 5.2**), I used estimated TSS and DIN loads (calculated from measurements close to river mouths) to calculate the percentage of TSS and DIN delivered by each river in relation to the total TSS/DIN load. I used annual loads for seven major rivers draining into the four selected NRMs to calculate their proportional contribution to the region-wide (only selected NRMs) pollutant load. End-of-system loads were estimated based on flow and water quality sampling in selected gauging stations (Joo et al. 2012) for five monitoring periods (2007-2011) for TSS and DIN (**Table 5.2**). The proportional contributions were calculated by dividing the TSS and DIN annual loads of each river by the summed load for all rivers.

River		2007	2008	2009	2010	2011
Barron	DIN	32	102	44	50	174
	TSS	30	397	163	174	239
Johnstone	DIN	518	409	487	339	731
	TSS	132	160	241	114	461
Tully	DIN	979	686	652	450	1,073
	TSS	116	88	113	70	232
Herbert	DIN	390	393	753	336	1,620
	TSS	337	220	1,888	336	1,344
Burdekin	DIN	1,815	2,646	2,324	1,036	2,588
	TSS	6,503	12,700	9,614	1,351	6,167
Pioneer	DIN	194	142	125	477	642
	TSS	156	255	111	374	819
Fitzroy	DIN	244	1,773	367	2,061	4,796
	TSS	320	4,751	404	3,564	8,665
TOTAL	**DIN**	**4,172**	**6,151**	**4,752**	**4,749**	**11,624**
	TSS	**7,594**	**18,571**	**12,534**	**5,983**	**17,927**

Table 5.2. Estimated annual end-of-system pollutant loads for major rivers of the GBR catchment in tons (DIN) and kilotons (TSS). Load estimates are based on measurements at gauge stations closest to river outlets. The figures underestimate loads because they do not include supply from coastal areas downstream of gauge stations, although the magnitude of the difference between estimated and total pollutant loads is unknown. Source: Queensland's Department of Environment and Resource Management.

Create annual grids of spatially distributed TSS and DIN loads for each river

In the fourth step, I created grids representing the annual average distribution of TSS and DIN delivered by the seven major rivers in the study region. I created grids for each river by multiplying their proportional contribution to the region-wide TSS and DIN loads (section 2.2.3) with a cost-distance grid defining the maximum area of influence and the dispersal of pollutants in the sea (described below). I then summed the individual spatially distributed grids (one per river) to represent the full TSS and DIN load per cell; the overlap of two or more grids defined cells influenced by multiple rivers.

I used ArcMap Spatial Analyst (ESRI 2010) to create the annual cost-distance grids for each river based on three inputs: source locations (points representing river mouths), a maximum distance (measured from river mouths), and a ñcostò grid (representing how easily the pollutants spread in the ocean). I calculated the maximum distance for each river based on the volume of the largest flood event of the year (which was assumed to produce the largest plume) and the estimated extents of studied river plumes. A review on the extent of the Burdekin plume indicates that the maximum northward distance reached by the plume from the river mouth is ~500km for extreme-very large discharge events (>12,000 GL) and ~200 km for average (3,000 to 7,000 GL) events (Lewis et al. 2006). I analysed daily hydrographs for each river (**Supplementary Material**) to identify flood events (defined as the sum of consecutive daily discharges exceeding the 95th percentile mean flow for a 10-year period: 2002-2011). Values over and under the 95th percentile marked the start and end of each flood event, and the largest events were those with the largest summed volume.

I calculated the maximum distance for each river/year using a power function that approximates the relation between the known distance (D) and volume (V) of the largest discharges (i.e., $D = X^{0.6}$) of the Burdekin river. I adjusted the distance when the observed extent (i.e., distance to the border of my mapped plumes) was larger than the distance predicted by this power function (**Table 5.3**). I calculated the cost surface (a grid defining the ñcostò per unit distance or impedance to move planimetrically through each cell) independently for TSS and for DIN. I did this using the grid representing the average colour-class value (i.e., the colour class most commonly found for each cell)

calculated from the weekly composites of 6-class plume maps. For TSS, a second polynomial function best fitted the observed relation between the 6 colour classes and nlw_667 (i.e., TSS proxy) and showed the expected gradient. For DIN I assumed conservative mixing (an approximation only) and thus used an inverse linear function that best fitted the relationship between colour classes and adg_443, a proxy for CDOM (CDOM has been used as proxy for salinity and thus to estimate area of riverine/plume influence; Schroeder et al. 2012). I normalised both DIN and TSS cost surfaces (**Figure 5.5.3**). The coastline and islands were assigned with óNo Dataô values, hence effectively acting as barriers in the calculation of the cost-distance surface and consequently influenced the modelled dispersal of pollutants.

River		2007	2008	2009	2010	2011
Barron	Volume	105	796	202	171	949
	Distance	**16**	**50 (65)**	**24 (75)**	**20 (75)**	**53 (106)**
Johnstone	Volume	370	470	800	208	547
	Distance	**35 (80)**	**40 (55)**	**55 (70)**	**25 (60)**	**44 (80)**
Tully	Volume	691	416	968	112	576
	Distance	**51 (80)**	**37 (70)**	**62 (95)**	**17 (75)**	**45 (115)**
Herbert	Volume	2,178	605	7,056	269	2,781
	Distance	**101**	**47 (80)**	**204**	**29 (90)**	**117 (130)**
Burdekin	Volume	6,260	14,849	22,443	1,401	9,495
	Distance	**190**	**318**	**408**	**77 (160)**	**244 (270)**
Pioneer	Volume	381	471	222	285	732
	Distance	**35**	**40**	**26**	**30**	**52**
Fitzroy	Volume	371	5,410	865	7,088	27,036
	Distance	**35 (150)**	**174**	**58 (150)**	**204 (300)**	**456**

Table 5.3. Volume of largest flood events and maximum distance of riverine influence. Values are estimates of the maximum linear distances (km) from river mouths that plumes can reach (and pollutants transported within) based on the relationship between discharge volume and observed extent of plumes. Distances in brackets are adjusted values based on observed extents of mapped plumes. Flow data were obtained from Queensland flow program: http://watermonitoring.derm.qld.gov.au/host.htm

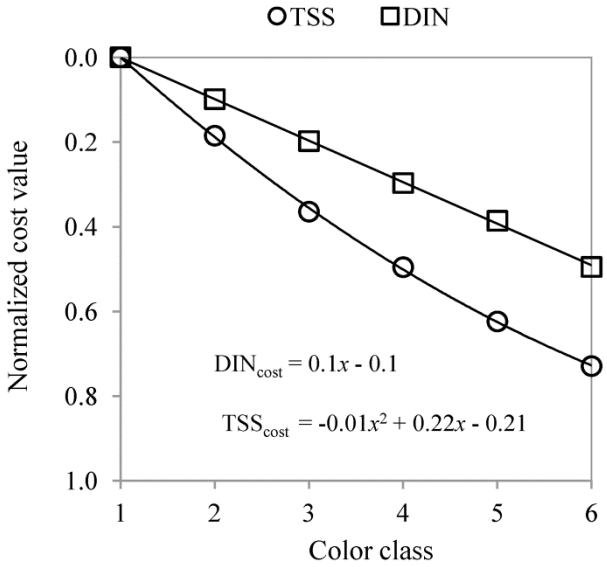

Figure 5.3. Functions used to calculate cost surfaces for total suspended sediment (TSS) and dissolved inorganic nitrogen (DIN).
These functions represent the best relationships obtained between colour classes and TSS and adg_443 (section 2.2.2). The line corresponding to TSS depicts the modelled decay function used to represent dispersal of sediments. The line corresponding to DIN indicates the estimated linear decay (assuming conservative mixing) of dissolved pollutants, in this case based on the observed behaviour of adg_443 (CDOM+D).

Calculate exposure by combining the frequency grid and scaled contributions

In the fifth step of the analytical process, to create the final exposure maps (E), I multiplied the annual frequency of plume occurrence grid (F) and the grid representing the sum of spatially distributed TSS and DIN loads for all rivers (P). I grouped exposure values in five categories of exposure (from very low to very high) to investigate spatial variation in exposure.

Spatial and temporal variability in exposure of marine ecosystems to plumes

Using the annual exposure maps, I quantified exposure to pollutants in two ways. First, I quantified the total area in each exposure class for TSS and DIN. Second, I calculated the area of coral reefs and seagrass beds affected by different exposure categories. I did this analysis for each year and plotted the variation in total area and affected areas of the

two studied ecosystems to analyses variation through time. I compared the total area and the affected area of the selected habitats under different exposure categories to understand inter-annual variability in exposure in relation to estimated loads. I also explored the relationship between the estimated maximum area covered by plumes and annual discharge.

Results

Modelling exposure

Plume mapping

Colour classes identified using my supervised classification method delineated the water masses and plume boundary visually recognisable in the true-colour images (**Figure 5.4**). The 6-class plumes showed the expected gradient in L2 parameters (**Figure 5.5**), and thus I considered the classes useful to qualitatively assess exposure to pollutants and to create exposure maps. Mapped plumes also provided enough detail to interpret the observed surface water masses. The first four colour classes (**Figure 5.4B**) included the more turbid plume zone, characterised by high TSS and light limiting conditions and very low salinity associated with the river mouth areas and the initial stages of plume formation. These four classes roughly match the primary water type (**Figure 5.4C**) described by Devlin et al. (2012b) and define the zone where most suspended sediments are deposited. The fifth class represented plume areas with moderately elevated TSS with sufficient light and excess nutrients to support elevated phytoplankton growth, similar to the secondary water type of Devlin et al. (2012b). The sixth colour class included riverine influenced waters, generally characterised by lower values of coloured dissolved organic and detrital matter (CDOM+D) than in previous classes, but still above ambient dry season values. This is similar to the tertiary water type of Devlin et al. (2012b).

The visual assessment of classified images indicated that using IHS images instead of RGB (not shown here) appreciably improved the correct delineation of the plume boundary and better discerned the very turbid brown water occurring near river mouths

and the less turbid transition zones, as well as better resolving the different intensities or hues of green water found within plumes further away from the coast.

Figure 5.4. Satellite image and mapped plumes for major rivers of the GBR, January 25th, 2011. A) MODIS true-colour image; B) 6-class plume map indicative of gradients in L2 parameters used in this study to assess exposure; and C) plume map depicting the three water types: primary (P), secondary (S) and tertiary (T) using the classification algorithm of Devlin et al. (2012b) based on ocean colour (L2) parameters. In B and C black areas indicate non-plume areas or portions of the image that were automatically masked out in SeaDAS (only for the PST map) in the processing of ocean colour parameters due to the presence of clouds or very turbid areas (resembling land).

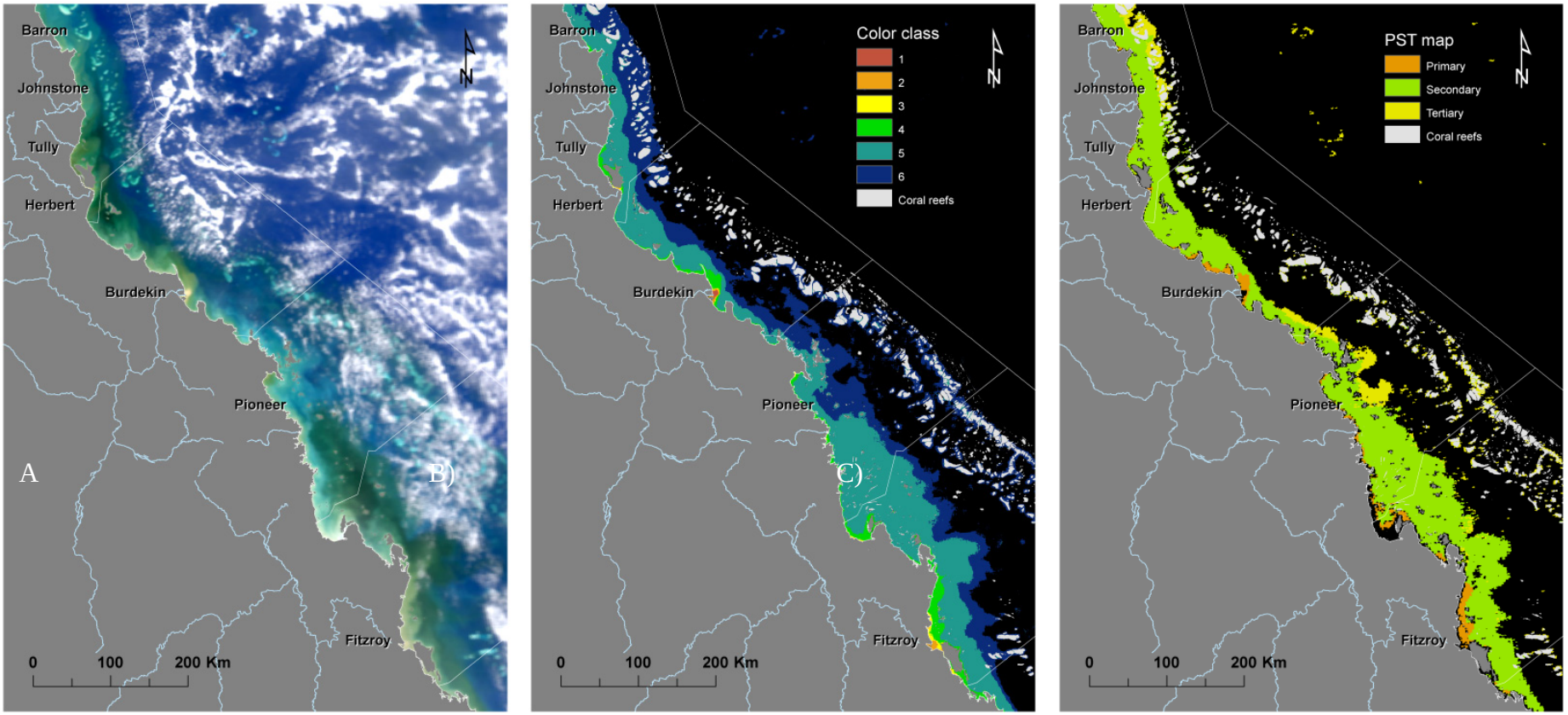

Figure 5.4. Satellite image and mapped plumes for major rivers of the GBR, January 25th, 2011

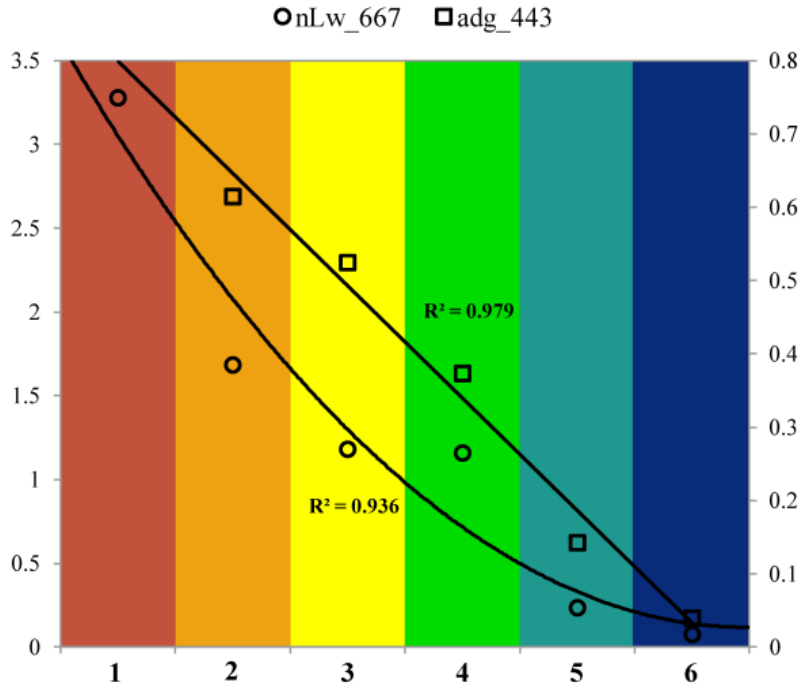

Figure 5.5. Variation in ocean colour (L2) parameters across the six colour classes identified using unsupervised classification.

Graphs summarise data from 25 selected images that cover different regions of the GBR with plumes of different size and with an observable gradient in surface colour classes (from very turbid brown to green water). Points are mean values for the two selected L2 parameters based on their recorded values within each colour class for the 25 images. The left Y-axis corresponds to the normalised water-leaving radiance at 667 nm (nlw_667), a L2 parameter used as a proxy for TSS. The right Y-axis indicates the variation in the CDOM and detritus absorption coefficient at 443 nm (adg_443), a parameter used as a proxy for variation in dissolved pollutants (e.g., DIN). Mean value for class 1 (very high TSS) of adg_443 (proxy for DIN) was excluded because the performance of standard algorithms for CDOM+D in highly turbid areas is low (Qin et al. 2007). The observed gradients observed in this graph were the basis for the functions used to calculate the cost surfaces for TSS and DIN (**Figure 5.3**).

Occurrence of plume and colour classes

Overlaying weekly composites of plumes showed the total area influenced by plumes and areas where plumes occur more or less frequently (calculate for each year independently), which approximates annual exposure to plumes (**Figure 5.6A**). The

mean value of colour classes (indicative of a gradient in L2 parameters; see **Figure 5.5**) corresponds to the grid (**Figure 5.6B**) used to represent dispersal of pollutants based on the functions described in **Figure 5.3**. Inter-annual variation in both frequency and mean colour classes (**Supplementary Material**) were reflected in the observed variation in exposure described in section 3.1.5.

Figure 5.6. Example of summary maps of plume occurrence used as inputs in the plume exposure model, in this case the images correspond to the 2011 wet season. A) annual frequency of plume occurrence (the dark red colour identifies areas influenced by plumes for most weeks during the wet season, while blue areas are indicative of areas rarely exposed to plumes); and B) mean value of colour class (see **Figure 5.5**), indicating the general pattern of pollutants dispersal during the wet season for that year. The colour coding is the same used in **Figure 5.5**. Black areas indicate non-plume areas.

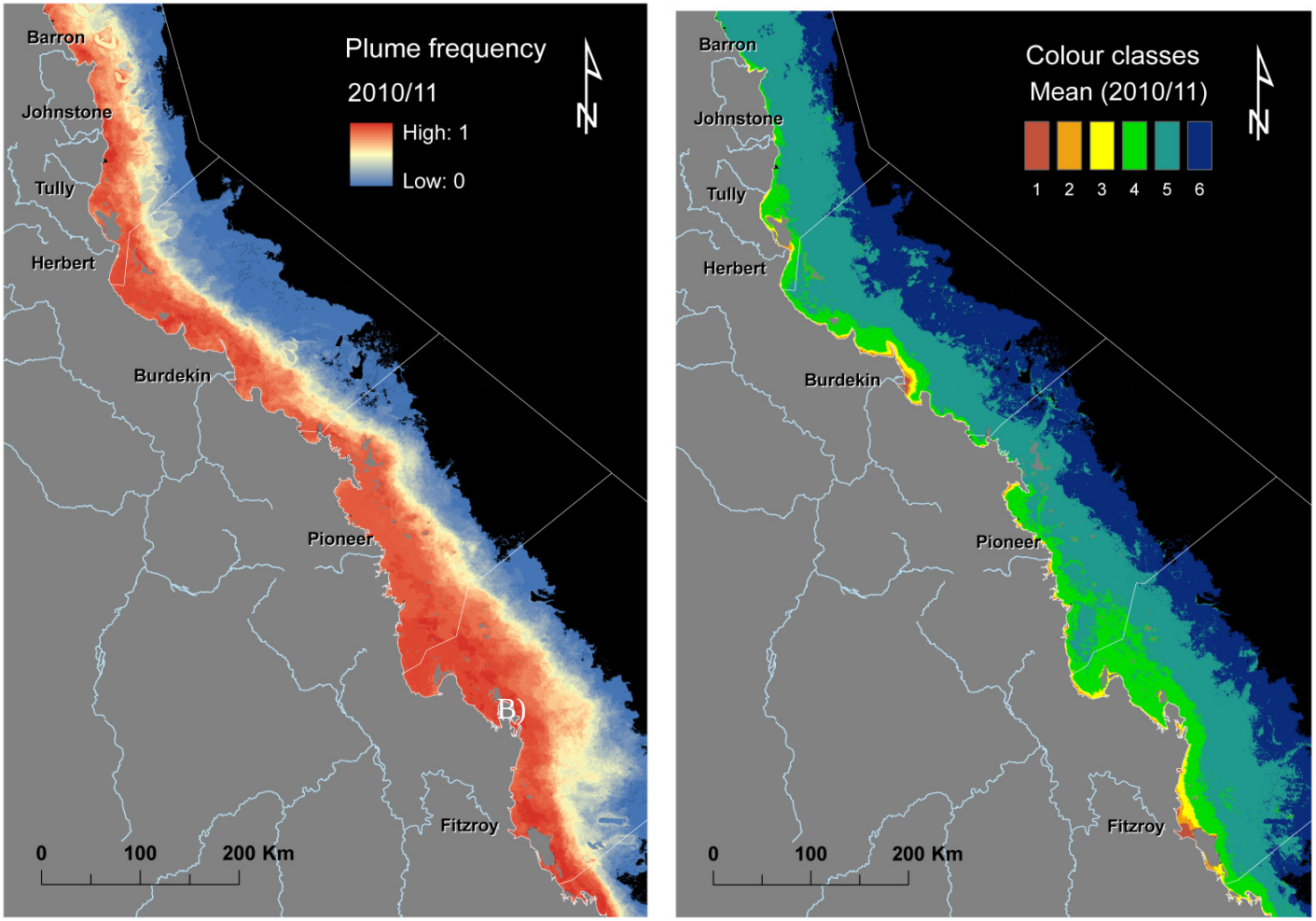

Figure 5.6. Example of summary maps of plume occurrence used as inputs in the plume exposure model

Proportional contribution of rivers

The delivery of DIN and TSS varied considerably between rivers, and relative delivery of rivers varied strongly between wet seasons (**Figure 5.7**). Overall, the two major rivers of the Dry Tropics (Burdekin and Fitzroy) contributed considerably to the region-wide DIN (49% to 72%) and TSS (80% to 94%) loads across all five wet seasons. Conversely, the summed proportional contribution of the Wet Tropicsôrivers (Barron, Johnstone, Tully and Herbert) was prominent for DIN (25% to 46%), but not for TSS (5% to 19%). Two rivers, the Herbert and the Tully, dominated the contribution of TSS and DIN loads in this NRM region. There was marked inter-annual variation in the proportional contributions of the two large Dry Tropics rivers (i.e., the average of the absolute deviations from the contribution of the Burdekin and Fitzroy rivers are 11% and 15% for DIN and 23% and 21% for TSS, respectively, while the average for all other rivers is only 2% and 1%). This pattern of delivery of pollutants indicates that areas under the influence of the two largest rivers are subject to more marked inter-annual delivery of pollutants, manifested in the patterns of exposure described below (sections 3.1.4 and 3.1.5).

Figure 5.7. Proportional contribution of major rivers to region-wide pollutant loads in each of the five wet seasons covered by this study. A) Dissolved inorganic nitrogen (DIN) and B) Total suspended sediment (TSS). Rivers identified by green colours in the bars correspond to rivers in the Wet Tropics NRM (**Figure 5.1**).

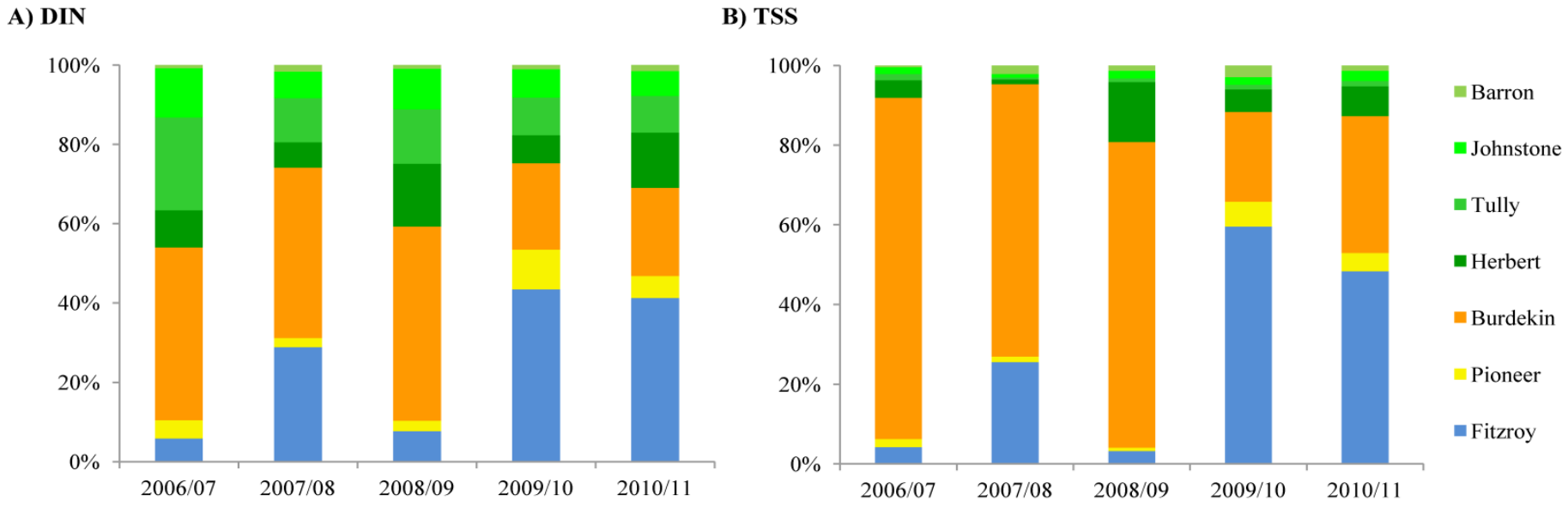

Figure 5.7. Proportional contribution of major rivers to region-wide pollutant loads in each of the five wet seasons covered by this study

Spatially distributed pollutant loads

The TSS and DIN cost surfaces (**Figure 5.8A, B**) ï used to represent differences in TSS and DIN dispersal and to create the grids of spatially distributed loads ï showed the expected spatial pattern of pollutant movement (Devlin & Schaffelke 2009), and reflected the exposure patterns to these pollutants (**Figure 5.8C, D**). Overall, I observed that the area estimated to be under major influence of DIN (identified as red to light green areas, **Figure 5.7**) is notably larger than that under influence of TSS, which reflects the expected rapid deposition of sediment in areas near river mouths (Bainbridge et al. 2012).

Figure 5.8. Cost surfaces and scaled loads for dissolved inorganic nitrogen (DIN) and total suspended sediment (TSS). A) Cost surface used to calculate the scaled (i.e., spatially distributed) loads of DIN presented in C for the 2011 wet season. The proportional increase in cost from the river mouths to offshore areas approximates the patterns of dispersal of DIN. B) Cost surface used to calculate the scaled (i.e., spatially distributed) loads of TSS presented in D for the 2010/11 wet season. Note the faster increase in cost relative to DIN. The patterns of the cost surfaces are reflected in the grids representing the scaled loads for DIN (C) and TSS (D), which can be interpreted as the probability of pollutants reaching a given area during the wet season of that year.

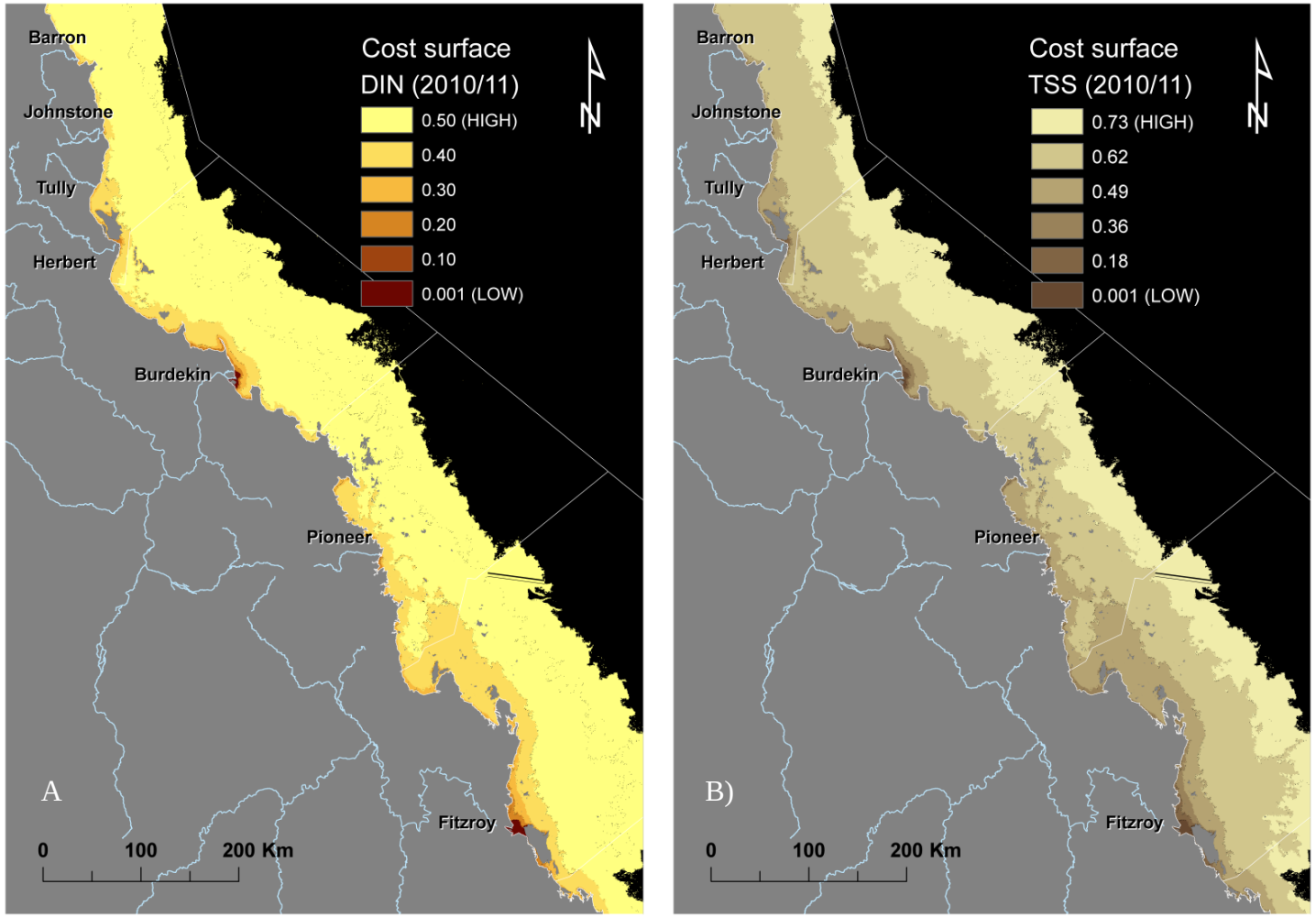

Figure 5.8. Cost surfaces and scaled loads for dissolved inorganic nitrogen (DIN) and total suspended sediment (TSS)

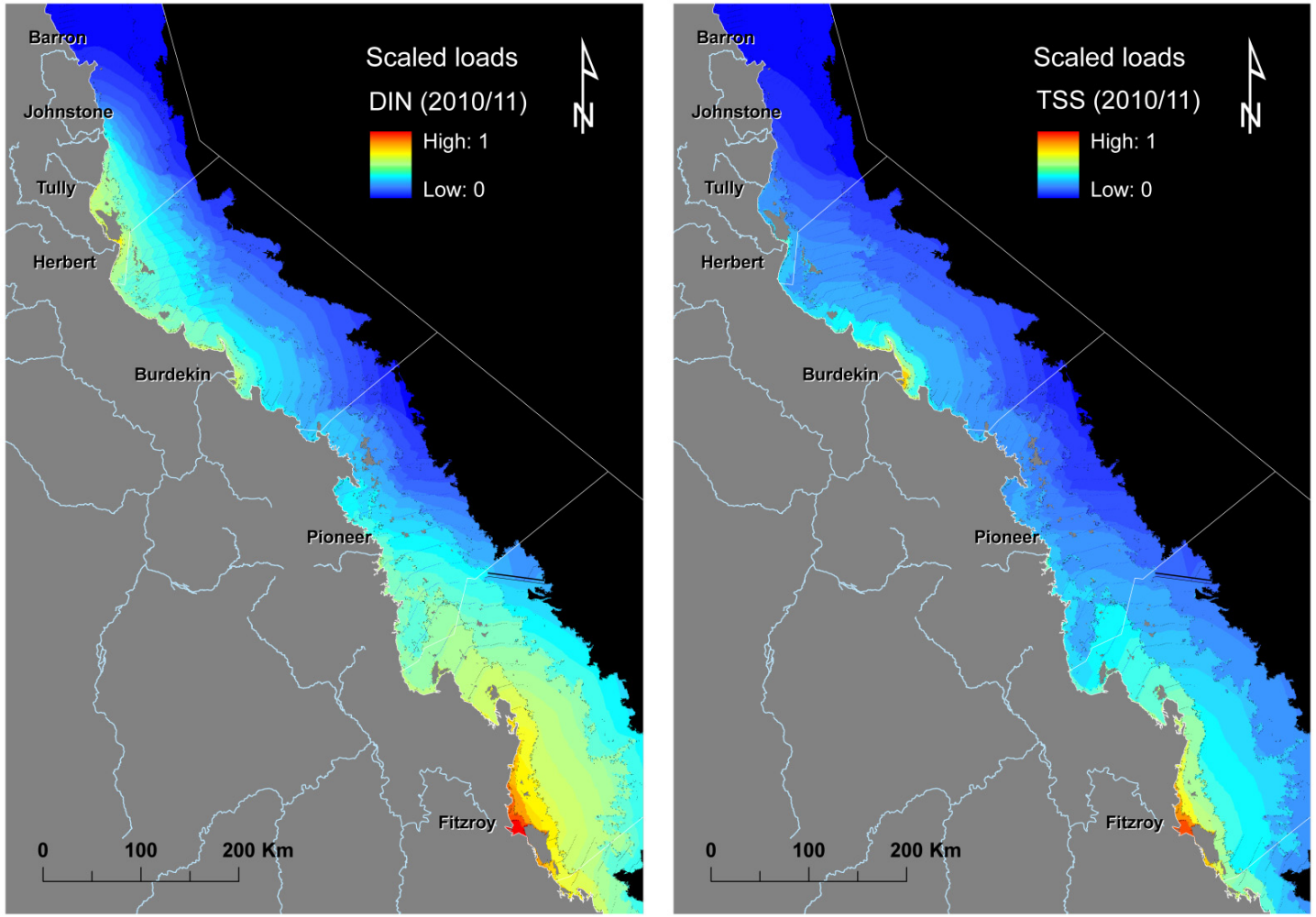

Figure 5.8. Cost surfaces and scaled loads for DIN and TSS (cont.)

Annual exposure to DIN and TSS

Coastal-marine habitats (coral reefs and seagrass beds) exhibited a range of exposures to DIN and TSS, reflecting the differences in the dispersal of pollutants and the locations of the habitats (**Figure 5.9**). The area under moderate to high exposure to DIN was larger than that for TSS across all studied wet seasons. Data for 2011, a wet season in which record discharges occurred, illustrated the degree of exposure that can be expected under extreme weather conditions (Devlin et al. 2012a), also manifested in the very extensive plumes within GBR coastal and offshore waters during this wet season (**Figure 5.10**).

Also worth noting are the differences in exposure of different habitats to DIN and TSS in response to the natural distribution of these habitats, as well as to the differences in the movement of pollutants. Overall, seagrass beds were commonly under higher exposure categories for both TSS and DIN, and most coral reefs were under low exposure categories, particularly for TSS.

Figure 5.9. Maps of annual exposure of coral reefs and seagrass beds to DIN (A) and TSS (B) for the 2011 wet season. The progression in intensity of colour indicates the degree of exposure to these pollutants. The distribution of seagrass beds in nearshore areas led to their higher exposure to both pollutants. Because of their proximity to the coast, coral reefs located within the Wet Tropics NRM are relatively more exposed to both pollutants in comparison with coral reefs of the Fitzroy NRM.

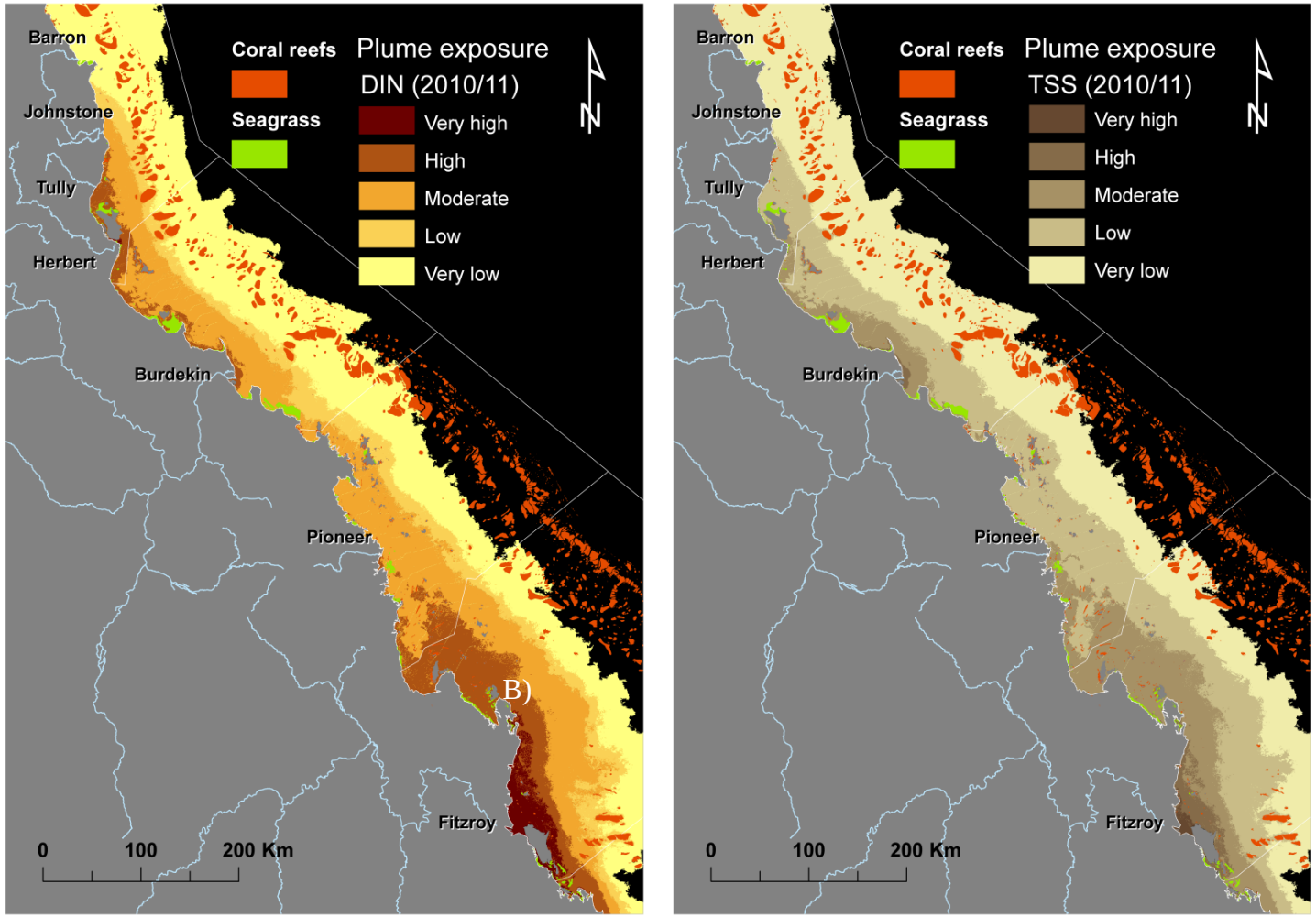

Figure 5.9. Maps of annual exposure of coral reefs and seagrass beds to DIN (A) and TSS (B) for the 2011 wet season

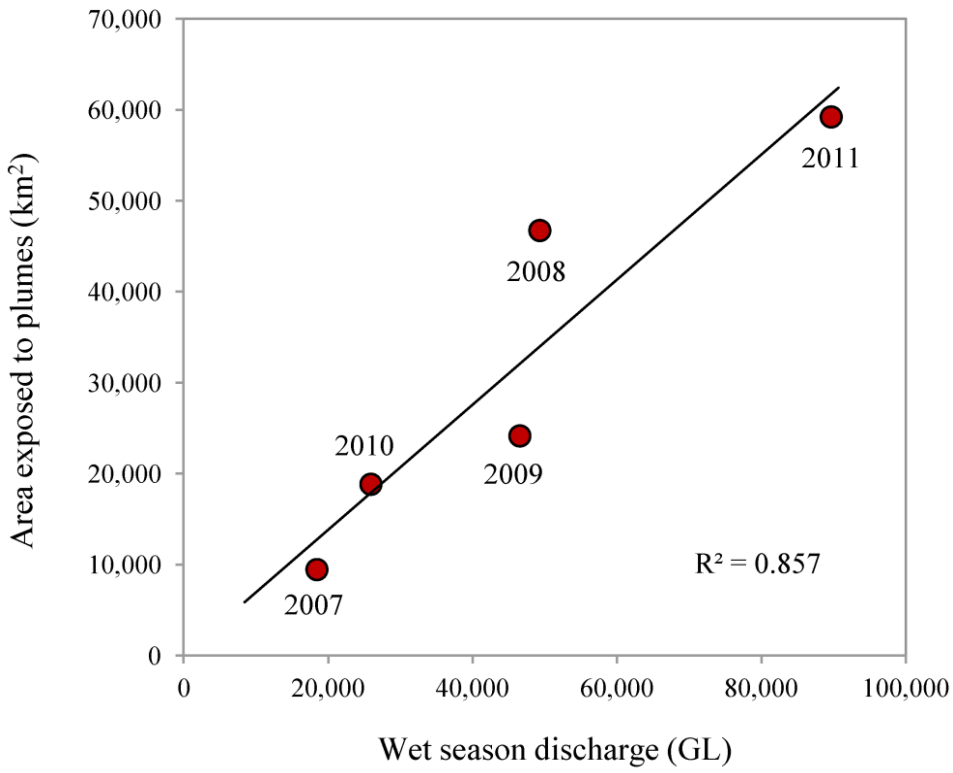

Figure 5.10. Relationship between modelled max. plume extent and recorded discharges.

Each point represents the summed discharge and the estimated area exposed to plumes for the wet season of that year. The graph shows a clear relation between discharge and the area of influence, and identifies wet years (e.g., 2008 and 2011), which were also characterised by large loads of TSS and DIN, and hence large areas under higher exposure categories.

Spatial-temporal variability in plume exposure

I found strong inter-annual variation in the area under exposure of the two modelled pollutants (**Figure 5.11)**. Differences in exposure were related to the sources of pollutants, and thus were strongly influenced by the proportional contributions of the different rivers to annual region-wide loads. The area of coral reefs and seagrass beds potentially affected by TSS and DIN exposure also varied considerably, with maximum numbers of coral reefs affected in 2011 (**Figure 5.12**). TSS exposure was high in both 2008/9 and 2010/11. While the exposed area can be large, the area influenced by high to

very high exposure categories was a small component for both DIN and TSS, with the exception of DIN in 2011.

Figure 5.11. Inter-annual variation in total area under different DIN/TSS exposure categories . A) Variation in exposure to DIN. This graph shows the large area under high exposure categories during 2011 (greatly influenced by the Fitzroy River: **Figure 5.7A**). B) Variation in exposure to TSS; in contrast to DIN, the largest area under high TSS exposure categories occurred during 2008 (in this case, largely driven by the Burdekin River: **Figure 5.7B**). In both graphs the horizontal bars connected by dashed lines correspond to the estimated loads of DIN (A) and TSS (B) for each wet season.

Figure 5.12. Inter-annual variation in exposure of coral reefs and seagrass beds to DIN (A and C, respectively) and TSS (B and D, respectively). The observed differences in exposure of habitats to TSS and DIN can be explained by their proximity to the coast, their location in relation to rivers contributing to the DIN load, and the estimated dispersal of both pollutants. In both graphs the horizontal bars connected by dashed lines correspond to the estimated loads of DIN (A and C) and TSS (B and D) for each wet season.

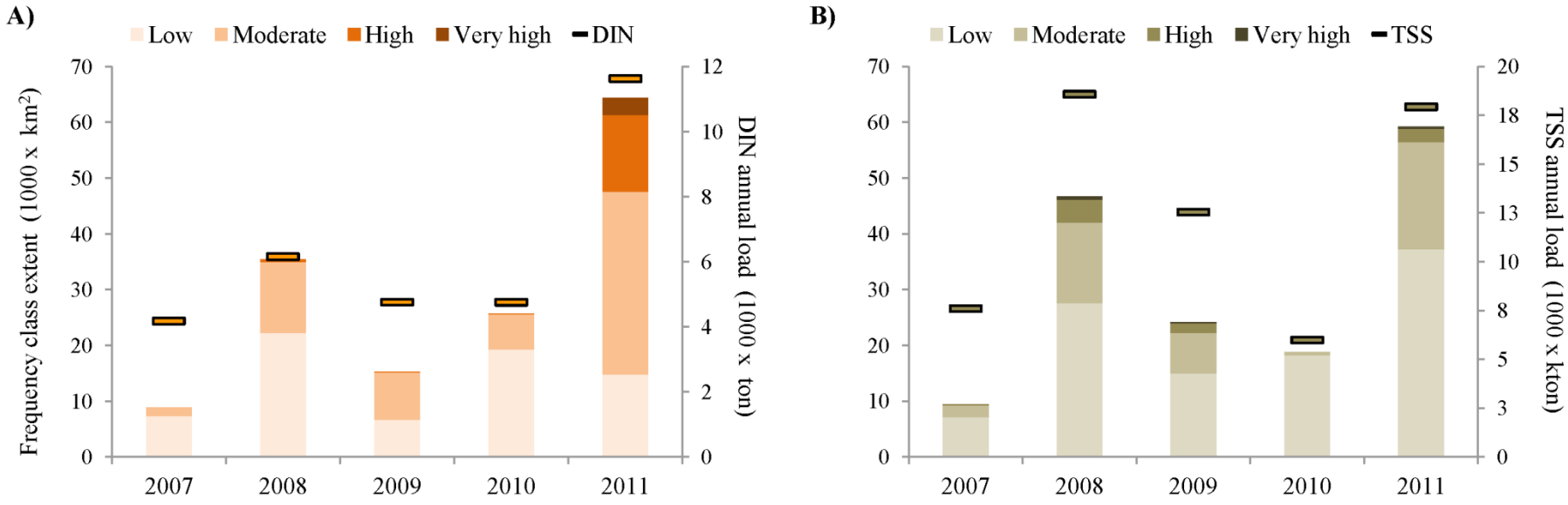

Figure 5.11. Inter-annual variation in total area under different DIN/TSS exposure categories

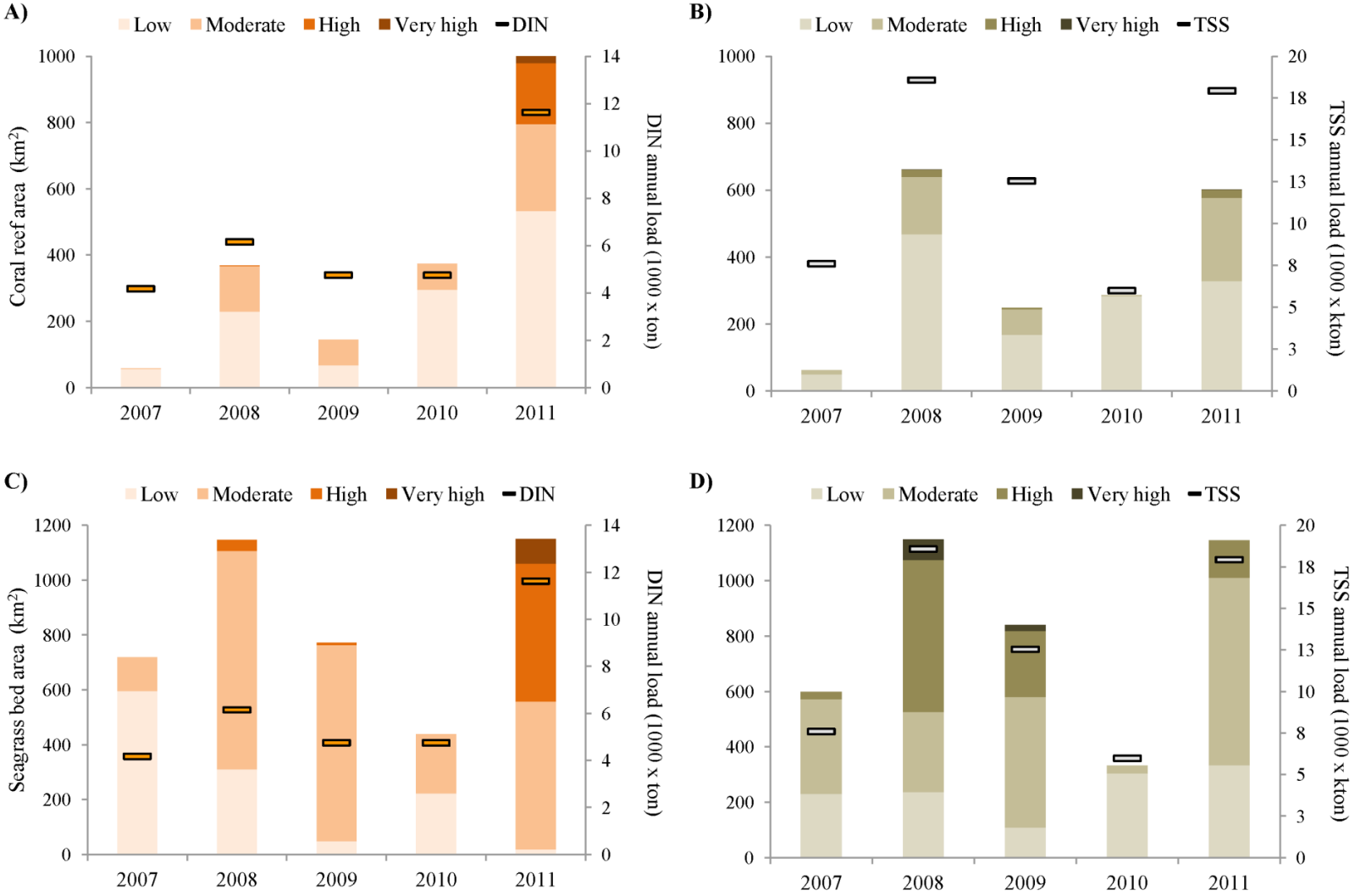

Figure 5.12. Inter-annual variation in exposure of coral reefs and seagrass beds to DIN (A and C, respectively) and TSS (B and D, respectively)

Discussion

Understanding the dynamics and potential impacts of land-based threats on coastal-marine ecosystems is a critical step towards minimising these impacts. My study made improvements to river plume exposure models in two ways: first, by developing a relatively simple method to map the extent and qualitatively assess dispersal of land-based pollutants; and second, by assessing spatial-temporal variability in the exposure of marine ecosystems to plumes. My work thus contributes to our understanding on plume spatial-temporal dynamics of the Great Barrier Reef and offers a method to improve plume exposure models developed for the GBR and elsewhere. The method is transferable to other regions, and can be scaled up or down to model the whole GBR, individual NRMs, or selected groups of catchments.

My plume exposure model incorporates some important improvements to previous models (Devlin et al. 2012b; Maughan & Brodie 2009). As stressed before, manual mapping of plumes has been used to model exposure, but is very time consuming, requires significant experience, and is not very precise. In addition, performance of methods based on ocean colour products are affected by atmospheric and water conditions. Classification of true-colour images allows more information to be retrieved from imagery (even under moderate cloud cover and sun glint conditions, or in very turbid areas). My approach also automates the mapping of images, thus reducing processing time and human error associated with visual interpretation of plumes. Manual mapping of plumes has been successfully used to model exposure, but is very time consuming and laborious, requires significant experience, and is not very precise. My exposure model further improves current methods by incorporating a simple way to integrate the dispersal of pollutants into river plume maps. In contrast, previous approaches applied a uniform value (representing the proportional contribution of an NRM to the region-wide loads) across for the whole NRM surface (Devlin et al. 2012b). An additional advantage of this approach is that exposure to pollutants throughout the GBR can be linked to specific rivers. This information is critical because, used in combination with maps of pollutant sources, it can guide targeting of catchment management to benefit particular habitats and parts of the GBR (Alvarez-Romero et al. In prep; Brodie et al. 2009). My method also allows the identification of areas likely

influenced by multiple rivers, hence incorporating a measure of cumulative impact. This relatively simple approach to mapping plumes and modelling exposure also facilitates the processing of large numbers of images, thus permitting the assessment of spatial-temporal variability across large regions and longer periods of time.

In the Great Barrier Reef, more accurate and consistent mapping of river plumes and derived exposure models could be important for monitoring land-based impacts and adjusting catchment management to minimise or mitigate those impacts. Land-based pollutants are recognised major stressors on coastal-marine ecosystems in the region (Brodie et al. 2012b), in particular for vulnerable habitats such as coral reefs (Fabricius 2011) and seagrass beds (McKenzie & Unsworth 2011). For instance, large-scale mortality events associated with waters of low salinity and high temperature caused by flood conditions have been documented for the regionôs coral reefs (Berkelmans 2009) and seagrasses (McKenzie & Unsworth 2011; Waycott et al. 2005). Flood plumes can result in low light levels in seagrass beds and coral reefs. Reduced exposure to light can cause lethal and sub-lethal conditions for photosynthetic organisms such as seagrass and coral symbionts (Collier et al. 2012; Cooper et al. 2008). Other long-term ecological impacts can be seen in the proliferation of coral-reef destroying crown-of-thorn starfish in areas regularly influenced by anthropogenic nutrient loads (Brodie et al. 2005; Fabricius et al. 2010). Recent seagrass mortality events associated with the 2011 wet season have also been related to increased mortality of dugongs and sea turtles in the GBR (McKenzie et al. 2011; McKenzie et al. 2012), highlighting the importance of studying variation in exposure to pollutants through time and its relation to observed ecological responses.

While my plume model was developed using the Great Barrier Reef as a case study, the approach is applicable to other regions, scales and pollutants. Although I expect that similar relationships between ocean parameters and plume characteristics can be found in other regions, calibration of functions used to model dispersal of pollutants is advisable because changes in concentration, composition, size distribution, shape, and refractive index of suspended, organic and inorganic marine particles affect the backscattering and absorption properties of water (Binding et al. 2005; Bricaud et al. 1995), hence its intrinsic colour. Information regarding loads was available for the study area through a long-term monitoring program (Carroll et al. 2012), but this information

may not be available for many regions. Under these data-limited circumstances the use of catchment models (Source Catchments, eWater_CRC 2010; e.g., SedNet, Wilkinson et al. 2004) to estimate pollutant loads is a good alternative (Alvarez-Romero et al. In review). The approach used to model exposure to DIN can be used for other dissolved pollutants (e.g., dissolved inorganic phosphorus, pesticides), thus extending the potential application of my model. However, the complex interactions between different pollutants (e.g., floculation: Bainbridge et al. 2012) and non-conservative mixing of some of these and other pollutants (i.e., their concentration gradient does not match that of salinity), such as particulate nutrients (Brodie et al. 2012a) warrants further investigation to adjust these models as necessary.

Some limitations are inherent in my plume modelling approach, some of which apply to other ocean colour remote-sensing applications. One aspect that requires careful consideration when applying the proposed model includes the presence of other surface water masses that resemble plume waters (see **Figure 5.4B, C**, where colour classes 5 and 6, and secondary/tertiary plumes are mapped in offshore reefs not associated with plumes). One example is the periodic upwelling of nutrient-rich water feeds colourful phytoplankton blooms, which can be confused with chlorophyll-rich plume waters (Beman et al. 2005). Another example is in areas with resuspension of terrigeneous fine sediments by currents, wind generated waves and tides in the period after floods (Fabricius 2011; Orpin et al. 1999). The latter situation can be observed in Shoalwater Bay, north of the Fitzroy River, where very turbid areas occur in the absence of plumes, mostly associated with strong tidal currents (Kleypas 1996). My model partially deals with these problems by focusing analysis on the wet season, masking out coral reefs and applying a cost-distance function, but further refinement could improve the model. Furthermore, the model might not accurately map plumes in shallow and narrow areas where bottom influence and radiance reflected back from adjacent land can contaminate signals recorded by the satellite (Doxaran et al. 2009; Ohde & Siegel 2001).

Conclusions

The maps resulting from the plume exposure model could be useful for a variety of applications, especially for monitoring of coastal-marine water quality and to prioritise

catchment management for marine conservation planning. This model, in combination with methods based on ocean colour parameters (Devlin et al. 2012b; Schroeder et al. 2012), has obvious applications for monitoring, because it could be a tool to periodically assess exposure of marine ecosystems to land-based pollutants. The model outputs are useful for conservation because understanding the dynamics and potential impacts of land-based threats is necessary to identify spatial alternatives that can minimise vulnerability and maximise conservation outcomes. These alternatives include avoiding imminent threats, where possible (e.g., by locating marine conservation areas away from areas strongly affected by land-based pollutants: Tallis et al. 2008), targeting catchment areas to minimise the supply of pollutants and improve end-of-river water quality (Brodie et al. 2009; Devlin et al. 2012b) and/or combining these strategies while assessing the trade-offs between them in different areas (Klein et al. 2010). To explore these options, planners require models of land-based threats that, ideally, integrate information on: i) the spatial distribution of pollutant sources; ii) the loads of land-based pollutants discharged into the sea; iii) the dispersal of these pollutants in the sea; and iv) the potential ecological impacts of degraded water quality. My model integrates the second and third elements, and can be linked to the first one through catchment modelling. Exploration of the correspondence of my exposure model to observed ecological changes (e.g., occurrence of COTS outbreaks, Fabricius et al. 2010; and declines in coral cover, Hughes et al. 2011; Sweatman et al. 2011), is yet to be investigated. Until now, incorporation of land-based threats in conservation planning applications is still limited (Alvarez-Romero et al. 2011), thus studies aimed to integrate these tools to better inform land-based threats models should be considered a priority.

Chapter 6. Advancing land-sea conservation planning: integrating land-use change, catchment and river plume modelling

Abstract

Human-induced disruption to flows of water, nutrients and sediments pose a significant threat to marine ecosystems. Ongoing land use change can further modify these flows, and amplify the impacts of land-based threats to vulnerable marine ecosystems. Consequently, there is an urgent need to assess these threats and prioritise actions to mitigate their impacts. A key question regarding prioritisation of actions is whether catchment management to maintain coastal-marine water quality can be spatially congruent with conservation actions for terrestrial biodiversity. In selected catchments draining into the Gulf of California, a marine biodiversity hotspot, I employed a land use change model to assess the vulnerability of areas with natural vegetation to conversion to crops, pasture and urban areas. I used catchment modelling to map the sources and estimate pollutant loads delivered to the Gulf by these catchments. I then prioritised areas for catchment management based on objectives for conservation of terrestrial biodiversity and objectives for water quality that recognised links between pollutant sources and affected marine areas, identified with a river plume model. My objectives for protection of terrestrial biodiversity covered species of vertebrates. My objectives for coastal-marine water quality were to reduce sediment and nutrient discharges from land uses and minimise increases in coastal sedimentation. I used conservation planning software (Marxan) to prioritise interventions and explore spatial differences in priorities for both objectives. Notable differences in the distributions of land values for terrestrial biodiversity and coastal-marine water quality indicated the potential for conflict and tradeoffs. However, there were spatial options to accommodate priorities to achieve both sets of objectives. My study demonstrates a practical approach to integrating models of catchments, land use change, and river plumes with conservation planning software to inform prioritisation of catchment management.

Introduction

Human-induced changes in flows of freshwater, nutrients and sediments are major threats to coastal and marine ecosystems worldwide (Gabric & Bell 1993; Tilman et al. 2001). Nutrient enrichment of coastal and marine waters associated with fertiliser use in agriculture creates eutrophic and hypoxic/anoxic conditions that affect marine ecosystems, biodiversity and human health (Diaz & Rosenberg 2008; Gabric & Bell 1993; Howarth 2008). Pesticides from agriculture are a growing concern in many regions, particularly for sensitive habitats and species such as mangroves, seagrasses, marine mammals, seabirds, and various fish and invertebrates (Duke et al. 2005; Haynes et al. 2007; Readman et al. 1992). Loads of sediments derived from land clearing, urbanisation, and agriculture have significantly increased in many regions (Walling 2006) and are a major threat to ecosystems, such as coral reefs and seagrass beds (Cabaco et al. 2008; Maina et al. 2011). Thus, regional studies to identify and assess land-based threats, and subsequently to explore options to mitigate these threats, are urgently needed.

Ongoing vegetation loss and land use change can further modify runoff and flows of nutrients and sediments (Hern§ndez-Guzm§n et al. 2008; Howarth 2008; Walling 2006), potentially magnifying current impacts of land-based threats to sensitive marine habitats (Cabaco et al. 2008; Reid et al. 2011; Rogers 1990). Worldwide, loads of suspended sediments and nutrients delivered to the oceans have increased drastically following extensive land clearing for grazing, agriculture and coastal development (Alvarez-Cobelas et al. 2008; Walling 2006). This highlights the need to better understand the magnitude and potential impacts of suspended sediments and nutrients on marine ecosystems, particularly in areas where land-based threats are prominent (Halpern et al. 2009; Maina et al. 2011). Alterations to pollutant loads resulting from land use changes have been estimated in some regions (e.g., Great Barrier Reef, Australia: Kroon et al. 2012), with a common finding being a dramatic (up to 10-fold) increase in pollutant discharges to coastal and marine environments. Our understanding of the drivers of land use change and ability to forecast these transitions have also advanced significantly (Aitkenhead & Aalders 2009; Kim 2010; Pfeifer et al. 2011). Integrating information from catchment models and land use change models is more recent (Burke & Sugg

2006; McColl & Aggett 2007; Paris & Cherubin 2008; Vanacker et al. 2003), and its application to marine conservation planning is urgently needed to better inform catchment management. A further level of integration with river plume models is desirable to relate sources of pollutants within catchments to specific marine ecosystems or recognised priority areas (Alvarez-Romero et al. 2011; Stoms et al. 2005).

Limited resources and other socioeconomic constraints require planners and decision-makers to prioritise catchment management and protection interventions to maximise downstream benefits. Methods to target catchment management have advanced rapidly in the past decade (Brodie et al. 2009; McKergow et al. 2005b; Meixler & Bain 2010), but integration of these into marine planning is uncommon (Alvarez-Romero et al. 2011). In the past 25 years, systematic conservation planning has developed principles, methods, and tools to guide conservation interventions that maximise benefits for biodiversity, while minimising the costs (Pressey & Bottrill 2009; Sarkar et al. 2006). Integrating land and sea is recent in this discipline, but some studies demonstrate the value of a systematic approach to prioritising catchment management to address land-based threats to marine ecosystems (Bohnet et al. 2011; Crist et al. 2009; Klein et al. 2010).

Given the urgency of minimising land-based threats to marine ecosystems, three key questions arise: how do sources of pollutants within catchments relate to specific coastal-marine ecosystems?, what are the different ways in which land-based threats to coastal-marine ecosystems can be minimised?, and, what are the similarities and differences in the distribution of land values within catchments for terrestrial and marine conservation? To answer these questions, I used the Gulf of California (GoC) as a case study, a marine biodiversity hotspot (Lluch-Cota et al. 2007; Olson & Dinerstein 2002), where multiple land-based threats have been documented (Lluch-Cota et al. 2007; P§ez-Osuna & Ruiz-Fern§ndez 2005), sometimes reaching offshore areas (Beman et al. 2005). A major gap in marine planning globally, including in the GoC, is the need to address land-sea connections, in particular land-based pollution, which has been identified as a key threat to marine ecosystems (Chapter 3).

The aim of this study is to integrate models of land use change, pollutant loads from catchments, and river plumes for land-sea conservation planning applications. In

particular, this study has two objectives. The first is to identify areas within catchments that need to be protected or managed to maintain or improve coastal-marine water quality, considering the influence of pollutants on specific marine ecosystems. Actions within catchments apply to areas with natural vegetation that could be protected against clearing to minimise further increases in sediment loads, and areas with human-modified land uses that require management to reduce current loads of pollutants delivered to the sea, for example through the implementation of best practices in agriculture and grazing. The second objective is to assess the spatial congruence between the maps depicting areas to protect for water quality (i.e., for marine conservation) and areas that are important for conservation of terrestrial biodiversity.

Methods

Several models and analyses were involved in my study. I used a land change model to estimate the probability of change from natural vegetation to anthropic land uses. I used a catchment model to identify areas prone to erosion if cleared (and thus potentially supplying more suspended sediments to coastal-marine areas) and to estimate potential reductions in pollutant loads (suspended sediment and dissolved inorganic nitrogen) if best management practices were implemented. I then combined the land change and catchment models to estimate the potential erosion resulting from changes in land uses. I also used a plume model to identify the marine areas potentially influenced by discharged pollutants, which I then linked to the sources of pollutants to be targeted for management. Finally, I used conservation decision support tools to identify and compare priorities for improving water quality (preventing erosion and reducing pollutant loads) and for conservation of terrestrial vertebrates.

Study area

I focused my study on the selected catchments of the Gulf of California, which drain into coastal-marine management units (**Figure 5.1**) identified as high priorities for conservation in national and regional contexts (SEMARNAT 2006). While the western coast of the Gulf of California remains comparatively undisturbed, many eastern coastal areas are affected by aquaculture, industry, agriculture, and urban areas (Lluch-Cota et

al. 2007). Hence catchments within this area are appropriate to study prioritisation of catchment management to achieve coastal-marine water quality objectives.

Figure 5.1. The selected thirty-nine catchments comprising the study area, all of which drain into the Gulf of California. A) Stream network and level of conservation priority of adjacent marine management units; B) Main land use classes, reflecting degrees of human modification (e.g., conversion of natural areas to cropland and pasture) associated with spatial patterns of pollutant supply.

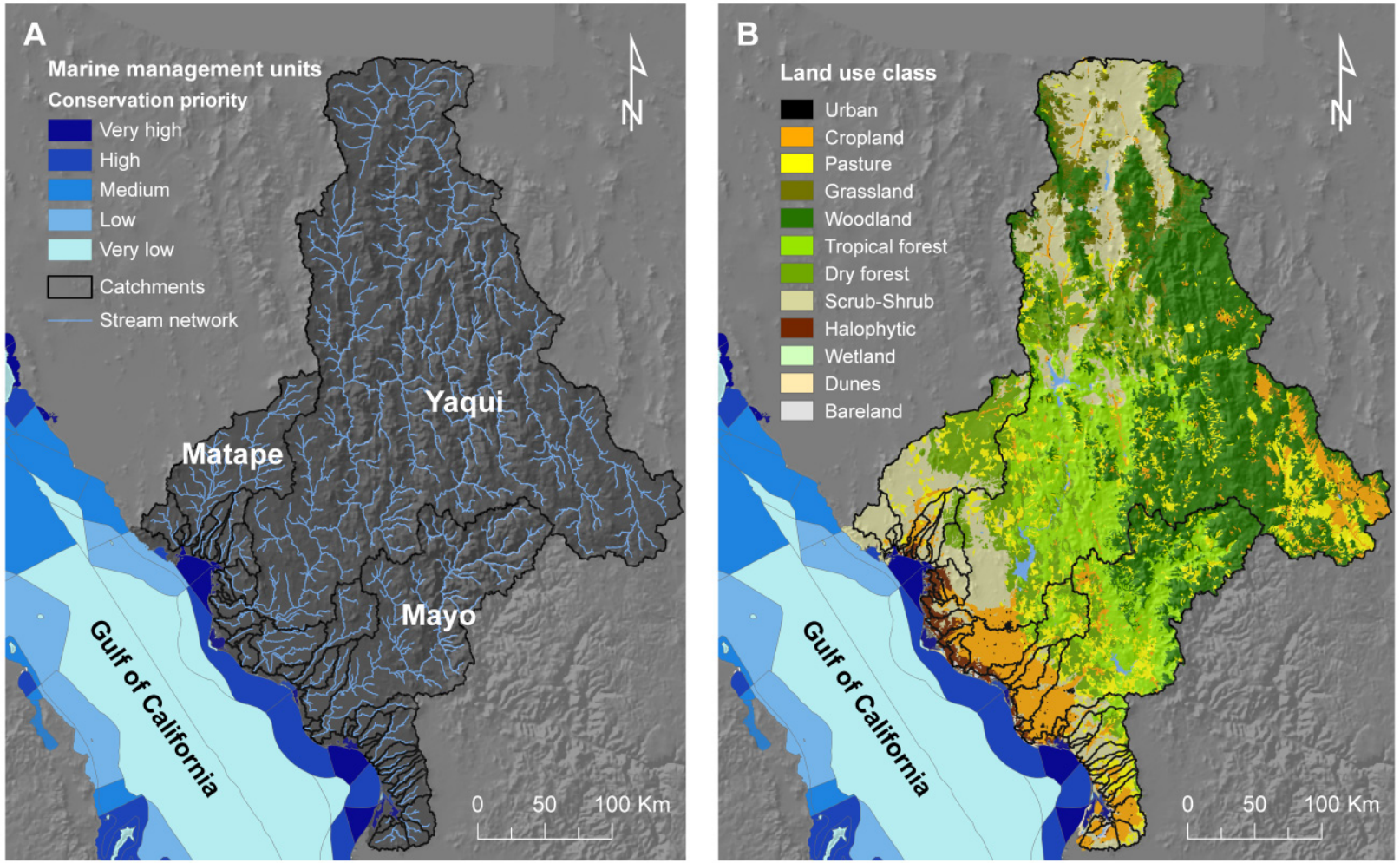

Figure 6.1. The selected thirty-nine catchments comprising the study area, all of which drain into the Gulf of California

Land use change modelling

I used a land use change model to estimate the probability of change from natural vegetated areas to cropland, pasture, and urban areas. This involved three steps: (1) exploring and summarising past change, which included correcting land-cover maps and selecting the datasets from the time series to be used in the land change model; (2) modelling transition potentials from natural vegetation to anthropic land use classes; and (3) calculating the probability of change for the modelled transitions within a predefined time period. These analyses were developed using ArcMap 9.2 (ESRI 2006) and Land Change Modeller (LCM, Eastman 2007), a tool commonly used for modelling of land use change (Kim 2010; Schulz et al. 2010).

I explored past changes in land use (step 1, above) by comparing three national-scale (1:250,000) land use/cover (hereafter óland useô) datasets: 1976, 1993 and 2000 (SEMARNAT-UNAM 2002), which were previously corrected for classification inconsistencies arising from different methods to develop them (Vel§zquez et al. 2002). I used LCM tools and GIS routines to identify and correct remaining class inconsistencies between time-series datasets. This process was needed to identify artificial differences between maps which could result in unlikely or false land use transitions (e.g., urban to rainforest, rainforest to temperate forest). I selected the 1976 and 2000 datasets as my reference maps to model land use change because false transitions for this pair of maps were less than those for the other two pairs (1976-1993 and 1993-2000) (Vel§zquez et al. 2002). I also created a map representing a ónatural scenarioô, defined as original vegetation cover before human modification. This map was constructed based on the earliest land use map available (i.e., 1976: SEMARNAT-UNAM 2002) for which anthropic land use classes were substituted using three reference maps: potential vegetation (Rzedowski 2006), ecoregions of North America (CEC 1997), and terrestrial ecoregions of Mexico (INEGI-INE-CONABIO 2007). This map was used to estimate the ónaturalô supply of sediments to identify the proportional increase in loads in comparison to the ócurrentô (2000) land use (**Figure 6.1B**).

To model transition potentials (step 2, above), I selected explanatory variables (including rainfall, dryness, land use, slope, soil type, and proximity to roads and developed areas) to model each of the transition potential sub-models. These variables

have been linked to land use suitability for human occupation and agriculture or related to physical accessibility in diverse regions (Kim 2010; Laurance et al. 2002; Rouget et al. 2003b). I created three transition potential sub-models, one for each anthropic class. I selected a set of explanatory variables for each sub-model. Only variables with Cramerôs coefficient V > 0.15 for each sub-model were incorporated. Cramerôs statistic has been used to test for strength of association (explanatory power) of variables in land use change applications (e.g., Almeida et al. 2005; Vanacker et al. 2003).

Finally, the selected variables were used to calculate the probability of change for the modelled transitions (step 3, above) within a 20-year period using a Multi-Layer Perceptron Neural Network (Chan et al. 2001). This is a useful technique to model change using multiple explanatory variables simultaneously, and for dealing with non-linear relationships (e.g., gradients in change probability with distance from developed or disturbed areas). I incorporated the influence of existing conservation areas (protection and management) in the model as óconstraintsô, thus effectively forcing LCM to reduce the probability of change from natural to anthropic land uses for areas covered by these two management tools. Based on expert advice, I assigned a high constraint value (0.25) to Protected Areas (CONANP 2010) and a low constraint value (0.75) to Wildlife Management Units (DGVS-SEMARNAT 2010). This resulted in three maps depicting the probability of change from all natural vegetation areas to each of the three anthropic classes (**Figure 6.2**).

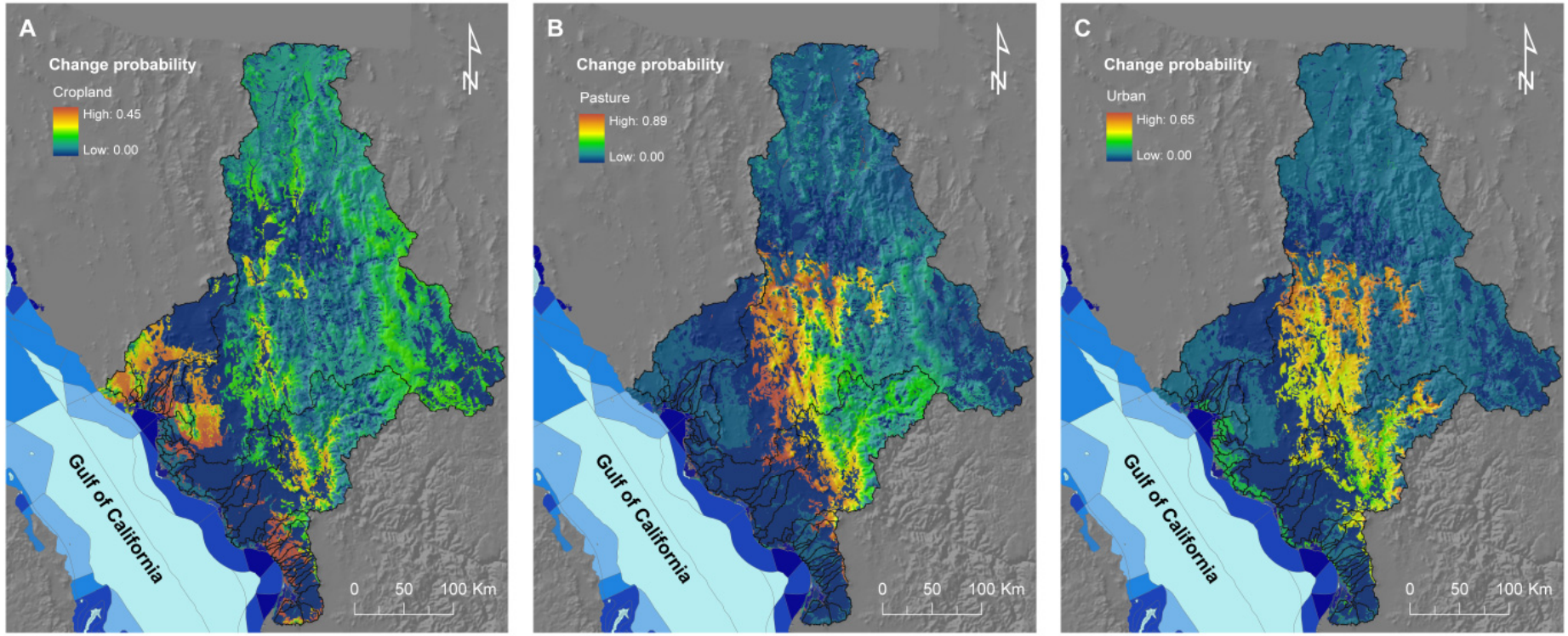

Figure 6.2. Probability of change from the eight natural vegetation classes to: A) cropland, B) pasture, and C) urban areas for a 20-year period (2000 to 2020) estimated using the land use change model.

Catchment modelling

I used SedNet (Wilkinson et al. 2004) to model supply (at subcatchment level) and total load (at catchment level) of fine total suspended sediments (TSS) from hillslope erosion, and nutrients (dissolved inorganic nitrogen ï DIN only). I modelled TSS to identify areas with natural vegetation that could be protected against clearing to minimise the increase in sediment loads delivered to coastal-marine areas. I also modelled DIN and TSS exported from cropland, pasture and urban areas to identify areas that need to be managed (e.g., through implementation of best-practices) to reduce pollutant loads discharged into the sea. I used SedNet because it predicts runoff and suspended sediment loads within their observed ranges of variation (Chapter 4). I delineated subcatchments with SedNet by using a drainage area threshold of 40 km^2, which resulted in 1,655 subcatchments (later used as planning units for the prioritisation analyses), with a mean size of ~60 km^2 (standard deviation: 39 km^2) and area/perimeter ratio of 1.1 (standard deviation: 0.44). The main requirements for delineated subcatchments related to catchment modelling (Chapter 4) and prioritisation analyses (see section 2.6).

I used the catchment model to create three types of outputs: i) total catchment loads for TSS and DIN based on ónaturalô and ócurrentô land use conditions; ii) reduction in TSS and DIN supply after implementing best-management practices; and iii) maximum TSS supply if existing naturally vegetated areas are transformed into anthropic land uses.

I calculated the total catchment loads (tonne/year) for TSS and DIN by adding the supply of their corresponding subcatchments, which were modelled using the ónaturalô and ócurrentô land use maps. Of the thirty-nine selected catchments draining into the marine management units, three catchments comprise 86% of the total study area (**Figure 6.1**): Yaqui (68,040km^2, 67%), Mayo (13,543km^2, 13%), and Matape (5,938 km^2, 6%), and combined contribute with approximately 96% and 94% of the ócurrentô TSS and DIN region-wide loads, respectively (Chapter 4).

To create the scenarios of potential reduction in TSS and DIN supply from implementing best-management practices I followed two steps. First, I created maps representing the estimated supply from subcatchments of DIN and TSS if 100% of the

areas under cropland, pasture and urban areas implemented best-management practices to reduce these two pollutants. To simulate reductions in DIN supply associated with best-management practices, I adjusted the event mean concentration ï EMC (see Bartley et al. 2011) of DIN for cropland and urban areas (**Table 1**). These values were set to represent maximum potential reductions in nutrient runoff based on reported reductions from literature (Wilkinson & Brodie 2011). Since potential reductions are higher for areas were fertiliser use is more intense, reduced EMC values differentially between cropland classes, depending on their relative use of fertiliser (i.e., very high use: 40% reduction; high use: 30% reduction; moderate use: 20% reduction; low use: 10% reduction; very low use: 5% reduction). For urban areas, the revised EMC value corresponds to a maximum 20% reduction. I excluded pasture from the DIN reduction scenario because pasture management practices have a minor impact on DIN concentrations. I simulated the effects of implementing best-management practices to reduce TSS supply by modifying the input into the catchment model known as the cover factor (C-factor is used in RUSLE to calculate soil loss: see Chapter 4) for cropland and pasture. I assumed a maximum TSS reduction of 30% from cropland and 20% from pasture, respectively (**Table 1**). I excluded urban areas from the TSS reduction scenario because, once these are established, they do not contribute significantly to erosion. Second, I calculated the difference between the óbest management practicesô and ócurrentô scenarios (described above) to represent a hypothetical maximum reduction (**Figure 6.3**).

Land use	Dissolved inorganic nitrogen (DIN)	Total suspended sediment (TSS)
Urban areas	0.794 (**0.635**)	0.005 (Not modified)
Cropland (5 classes)	Very highé 1.500 (**0.900**) Highé é ...1.047 (**0.733**) Moderate....0.850 (**0.680**) Lowé é ....0.750 (**0.675**) Very lowé .0.700 (**0.665**)	0.261 (**0.183**)
Pasture	0.399 (Not modified)	0.230 (**0.184**)

Table 1. Nutrient event mean concentrations (mg/L) and C-factor (non-dimensional parameter) values (for DIN and TSS, respectively) used for catchment modelling. The first set of values correspond to the original EMC and C-factor values used to model ócurrentô and maximum supply scenarios. Numbers in parenthesis and bold are the modified parameters used to simulate TSS and DIN reductions resulting from implementing best-practices. I classified cropland areas based on their relative use of fertiliser, from very low to very high (SEMARNAT-INE 2000) and used this classification to progressively assign EMC values for each class (lowest to highest) using the 50th, 60th, 70th, 80th, and 90th percentiles of documented values for cropland (**Supplementary Material**).

Figure 6.3. Estimated maximum reduction in the supply of pollutants (total suspended sediment: TSS and dissolved inorganic nitrogen: DIN) from anthropic land uses after implementing hypothetical best-management practices. The maps show the maximum reduction (kg/year) in TSS supply (at subcatchments level) for A) cropland and B) pasture; and DIN for D) cropland and D) urban areas. Only subcatchments including anthropic land uses were targeted for management in the prioritisation analyses based on best-practice land uses.

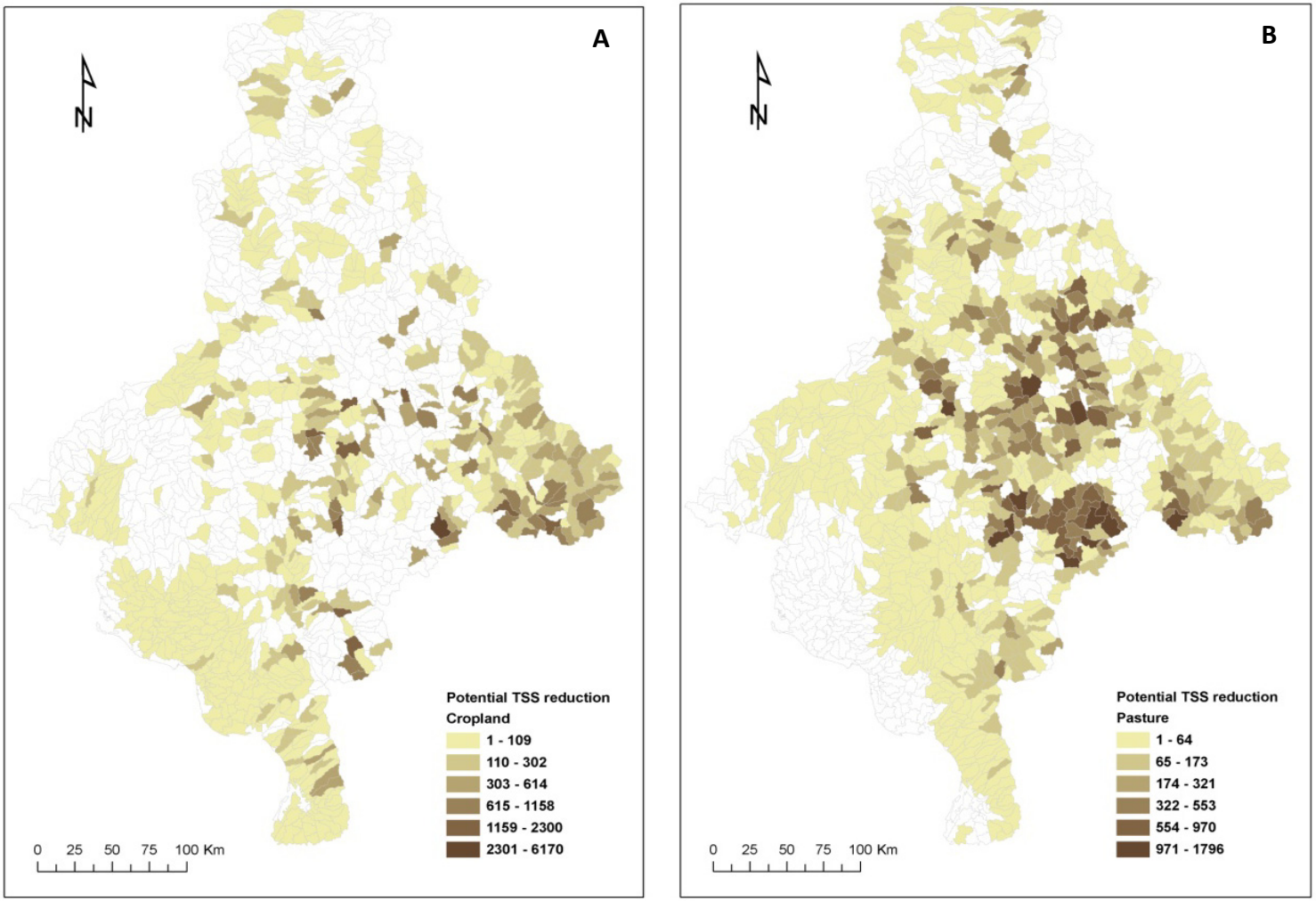

Figure 6.3. Estimated maximum reduction in the supply of pollutants (total suspended sediment: TSS and dissolved inorganic nitrogen: DIN) from anthropic land uses after implementing hypothetical best-management practices

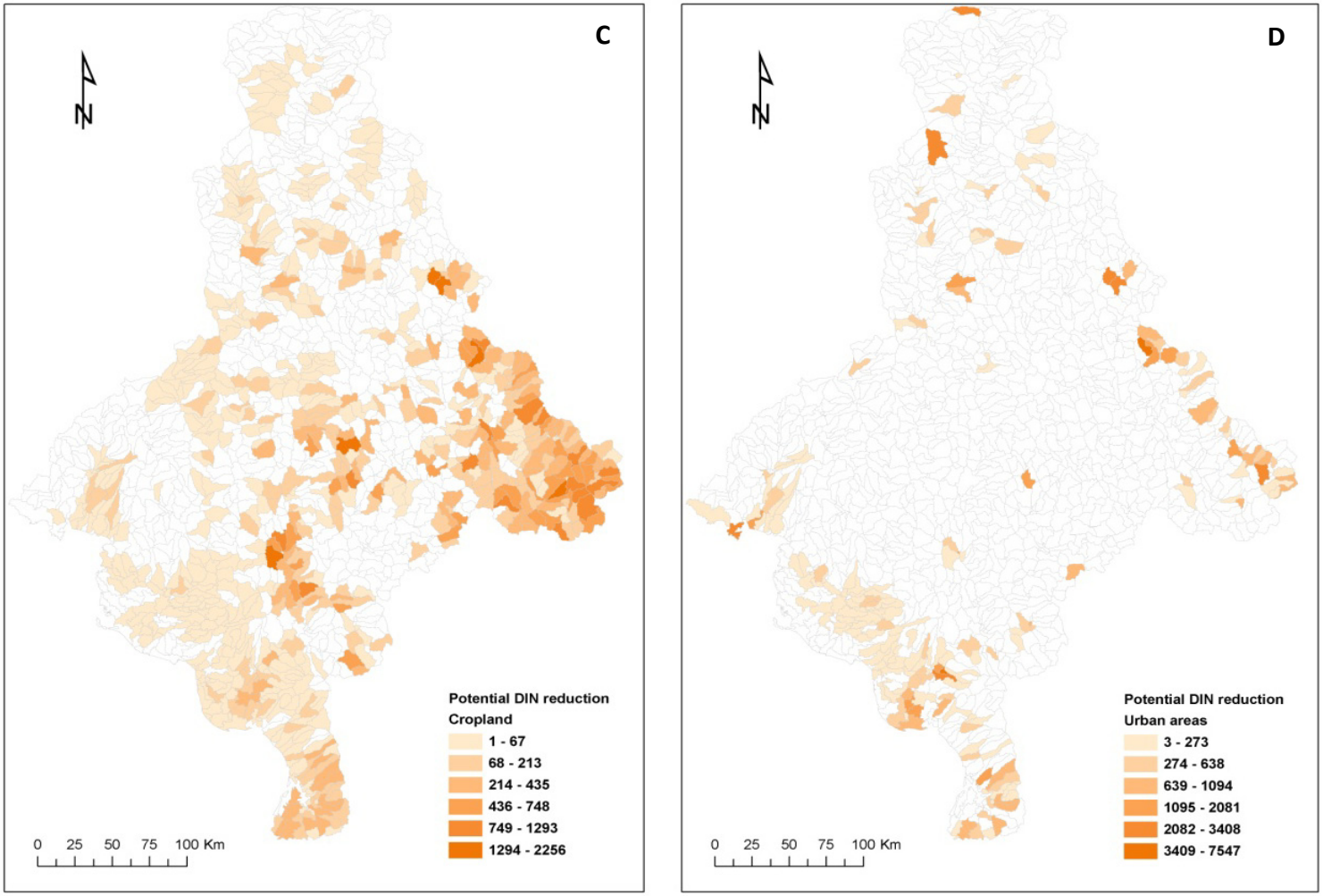

Figure 6.3. Estimated maximum reduction in the supply of pollutants (total suspended sediment: TSS and dissolved inorganic nitrogen: DIN) from anthropic land uses after implementing hypothetical best-management practices

Finally, I modelled the maximum TSS supply assuming all remaining naturally vegetated areas are transformed into anthropic land uses. I created these maps by applying a uniform C-factor (one for each of the modelled transformation: cropland, pasture or urban areas) to all areas with any of the eight natural vegetation classes.

Combining land use change and catchment models

The outputs from land use change and catchment models were combined to create a map representing the estimated proportional contribution of subcatchments to the region-wide TSS load if natural vegetated areas were transformed into anthropic land uses. I focused on TSS because I only modelled probability of change (i.e., soft prediction) to anthropic land uses and not the future distribution of land uses (i.e., hard prediction), which would be required to model increases in DIN loads. To do this, I first transformed the maps showing the estimated maximum TSS supply assuming all areas with natural vegetation are converted into cropland, pasture or urban areas into maps depicting the proportional contribution (0 to 1) of each subcatchment (only subcatchments with natural vegetation) to the region-wide (study area) TSS load. Then, I combined each of the transformed maps (i.e., proportional maximum contribution from each transition: natural-to-cropland: $TSS_{crops\text{-}max}$, natural-to-pasture: $TSS_{pasture\text{-}max}$, and natural-to-urban: $TSS_{urban\text{-}max}$) with the maps depicting the probability of change from natural vegetation to anthropic classes (one per transition: natural-to-cropland: P_{crops}; natural-to-pasture: $P_{pasture}$; and natural-to-urban: P_{urban}) calculated using the land use change model (section 2.2). This created a map representing the integrated maximum proportional contribution ï $TSS_{integrated\text{-}max}$ (**Figure 6.4**) based on the following function:

$$TSS_{integrated\text{-}max} = (P_{crops} \text{ Ł} TSS_{crops\text{-}max}) + (P_{pasture} \text{ Ł} TSS_{pasture\text{-}max}) + (P_{urban} \text{ Ł} TSS_{urban\text{-}max})$$

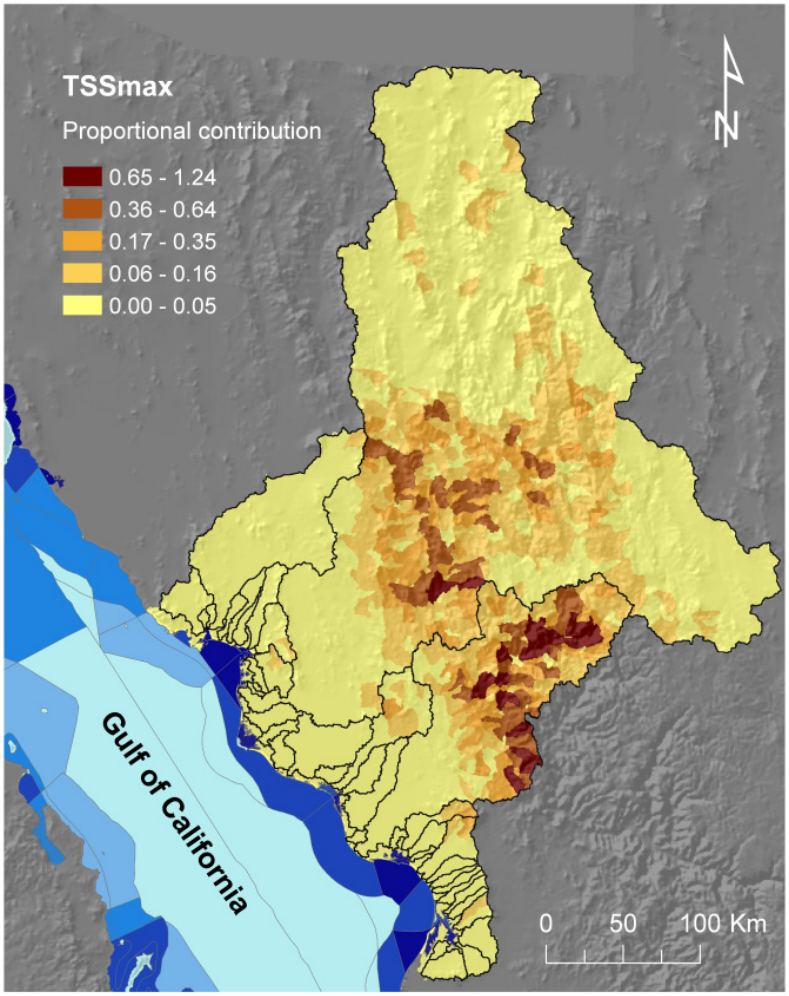

Figure 6.4. Maximum proportional contribution of subcatchments to the region-wide suspended sediment (TSS) load, considering the potential conversion of natural vegetation to cropland, pasture, and urban areas.

The map integrates the individual maps depicting the potential supply of TSS if natural vegetated areas are converted into cropland, pasture or urban areas (**Figure 6.3**), each previously multiplied by the probability of change from natural vegetation to the corresponding anthropic land use (**Figure 6.2**).

River plume modelling

I used a global model for land-based threats (Halpern et al. 2008) as an approximation of river plumes in the study area. As part of their map of cumulative land-based impacts, Halpern et al. (2008) modelled marine areas likely influenced by nutrient inputs (**Figure 6.5A**). I used the link between catchments (i.e., sources of land-based pollution) and marine areas (i.e., marine management units) to modify the features

targeted to achieve water quality objectives (for both management and protection) based on the priority of marine management units (described below). Due to the lack of spatial information on the extent of river plumes originating from the smaller coastal catchments, I assumed that their influence would be restricted to marine management units into which they drain directly.

I used the model of Halpern et al. (2008), in combination with the priority level of marine management units (Figure 1, SEMARNAT 2006), to calculate a marine priority (MP) factor. The MP factor represents the importance of subcatchments for coastal-marine water quality (**Figure 6.5B**) based on the priority of marine management areas that are potentially affected by the catchment to which each subcatchment belongs (section 2.4). Subcatchments that are linked to high conservation priority marine units have a higher MP value. I calculated MP using the following function:

$$MP = CI \cdot VI$$

Where

CI: Conservation Index (0 to 1): ecological importance based on species richness, productivity, presence of threatened/protected species (e.g., *Phocoena sinus*, *Totoaba macdonaldii*, sea turtles, cetaceans, sharks), endemic algae, wetlands, bays and coastal lagoons, and protected areas.

VI: Vulnerability Index (0 to 1): Higher VI values indicate areas where high fragility and human-pressure values co-occur; these areas should be considered by federal and state governments as a priority for conservation/management.

Both indices (CI and VI) were allocated to marine management units following a consultative process with experts and stakeholders as part of a marine spatial planning process for the Gulf of California (SEMARNAT 2006).

Figure 6.5. River plume model and priority level of catchments for coastal-marine conservation. A) River plume model depicting potential impacts from land-based nutrient pollution (Halpern et al. 2008) used as a proxy of the extent of influence of the

selected rivers. B) Marine priority (MP) factor assigned to each catchment (and subsequently to the subcatchments therein), representing their importance for marine conservation in terms of their link to marine management units of varying conservation priority.

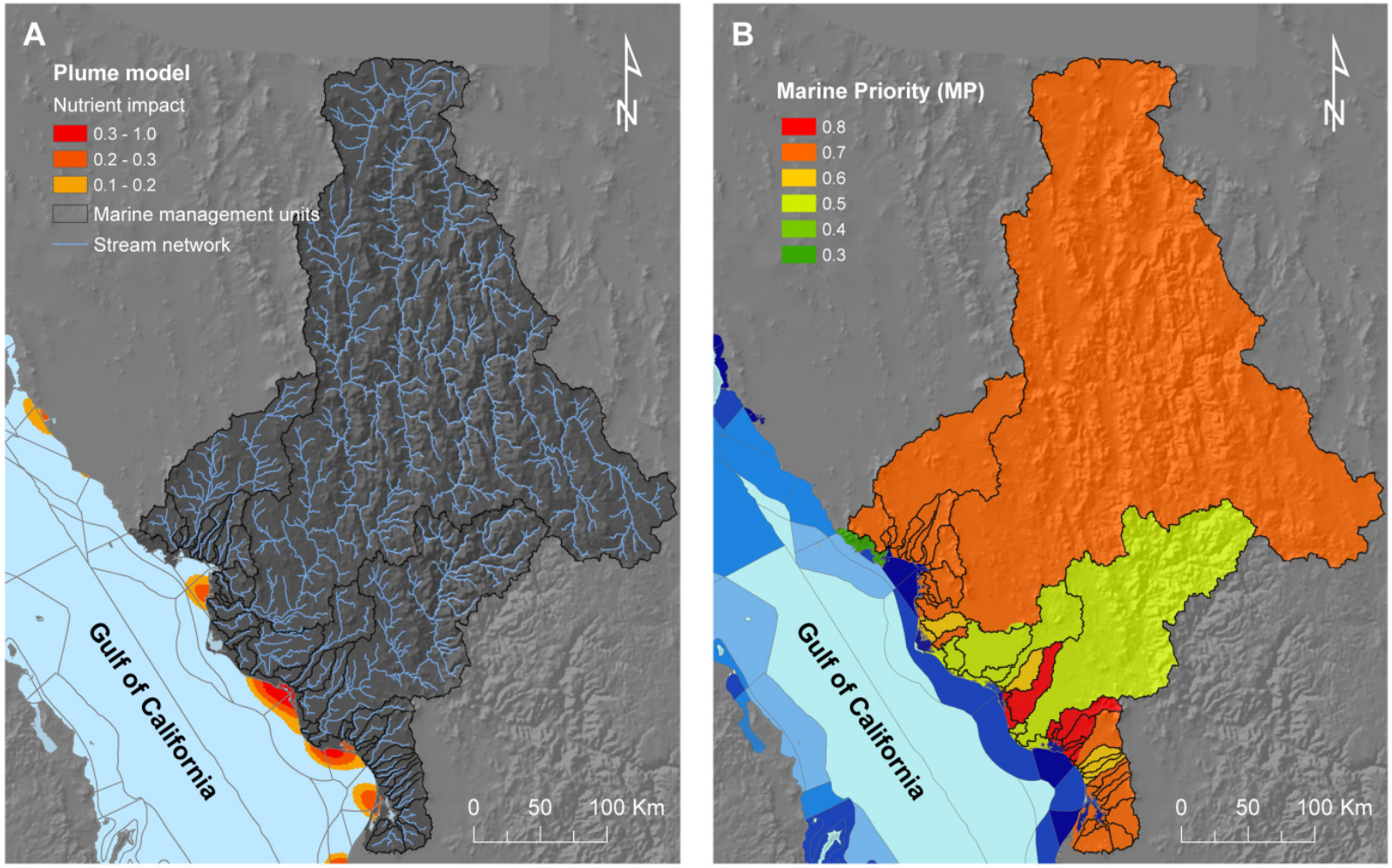

Figure 6.5. River plume model and priority level of catchments for coastal-marine conservation

Management scenarios and objectives

I defined two general management scenarios to guide my prioritisation: i) implementation of best practices in anthropic areas to improve coastal-marine water quality; and ii) protection of natural vegetation to maintain coastal-marine water quality and to conserve terrestrial biodiversity (**Table 2**). Both scenarios incorporate objectives to minimise land-based pollution to priority marine conservation areas. I used the decision-support tool Marxan (Ball et al. 2009) to explore the management scenarios. I used the modelled subcatchments (as delineated in SedNet) as the planning units. These subcatchments were selected particularly to make the total number of units analytically tractable (the number of analysis units strongly influences processing time), and to vary the size of units (important because I used area as a proxy for cost in Marxan scenarios). Below I describe the scenarios and how I developed features and objectives for use in Marxan; additional details about my use of Marxan are given in the caption for **Table 2**.

Scenarios	Goal	Description	Features targeted	Objectives
1. Management of anthropic land uses	Improve coastal-marine water quality through best practices (WQ1)	Implement best practices in areas with anthropic land uses to reduce loads of nitrogen and sediment delivered to priority marine conservation areas	Modelled maximum reduction of DIN (from cropland and urban areas) and TSS (from cropland and pasture)	Arbitrarily set as 30% of the estimated TSS and DIN potential reduction from the corresponding anthropic land use
2. Protection of natural vegetated areas	Maintain coastal-marine water quality through protection (WQ2)	Protect areas with natural vegetation that are prone to erosion and likely to be converted into anthropic land uses to avoid increases in sediment delivered to priority marine conservation areas	Modelled maximum supply of TSS from subcatchments with remaining natural vegetation	Arbitrarily set as 30% of the maximum TSS supply from natural areas if converted to anthropic land uses
	Conserve terrestrial biodiversity	Protect areas with remaining natural vegetation that are important for the conservation of vertebrate species of conservation concern	Modelled distributions of endangered and protected species of terrestrial vertebrates	Set as variable percentages of potential distributions; objective for each species determined with expert opinion

Table 2. Summary of parameterisation for Marxan prioritisation scenarios. Each scenario used area as the cost (scaled by a factor of 100), a uniform feature penalty factor (10), and BLM set to 0 (no clumping). Each scenario was run 100 times, with 1,000,000 iterations each.

Scenario 1: management of anthropic land uses

Water quality objective 1: improve coastal-marine water quality through best practices

The first set of water quality objectives (WQ1) considered the implementation of best practices in areas with anthropic land uses (cropland, pasture and urban areas) to reduce loads of DIN and TSS delivered to priority marine conservation areas (**Table 2**). I set four objectives for end-of-river loads of TSS and DIN (two for DIN and two for TSS) based on feasible reductions following the implementation of best practices (section 2.3). I arbitrarily set these objectives as a 30% of the total potential reduction following implementation of best practices in cropland and pasture (for DIN), and cropland and urban areas (for TSS). I used the hypothetical maximum reduction (per subcatchment) to prioritise management by targeting subcatchments that contribute the most to the region-wide loads of DIN and TSS. I incorporated the link to marine areas by scaling the four maps depicting maximum potential reduction ($TSS_{crops\text{-}reduction}$, $TSS_{pasture\text{-}reduction}$, $DIN_{crops\text{-}reduction}$, and $DIN_{urban\text{-}reduction}$, **Figure 6.3**) using two factors (calculated at catchment level): marine priority (MP: section 2.5) and proportional change (change factor) in total TSS and DIN loads (CF: described below) using the following functions:

$$TSS_{crops\text{-}scaled} = TSS_{crops\text{-}reduction} \cdot MP \cdot CF_{TSS}$$
$$TSS_{pasture\text{-}scaled} = TSS_{pasture\text{-}reduction} \cdot MP \cdot CF_{TSS}$$
$$DIN_{crops\text{-}scaled} = DIN_{crops\text{-}reduction} \cdot MP \cdot CF_{DIN}$$
$$DIN_{urban\text{-}scaled} = DIN_{urban\text{-}reduction} \cdot MP \cdot CF_{DIN}$$

The change factor (CF) represents the proportional change in TSS load from ónaturalô to ócurrentô land use conditions, and is calculated at catchment level (**Figure 6.6**). This factor was incorporated to increase the importance (i.e., increasing proportional contribution) of catchments that have experienced more change in the amount of exported sediments and that are, at least potentially, having a higher impact on marine ecosystems. I assumed that relative changes in DIN and TSS loads are reflected in relative changes in their impact on coastal-marine ecosystems. I assigned the CF to each catchment (and units within) based on the observed proportional changes in loads of TSS and DIN from ónaturalô to ócurrentô land use according to **Table 3**.

Dissolved inorganic nitrogen (DIN)	Suspended sediment (TSS)	Change Factor (CF)
< 25%	< 25%	0.5
25 - 100%	25 - 75%	0.6
100 - 250%	75 - 100%	0.7
250 - 500%	100 - 200%	0.8
500 - 1000 %	200 - 350%	0.9
> 1,000%	> 350%	1.0

Table 3. Ranges of proportional change in catchment loads of pollutants (DIN and TSS) from ónaturalô to ócurrentô land use conditions used to calculate the change factor (CF). I arbitrarily applied progressively larger CF values, from 0.5 (i.e., reducing the importance of the catchment by half) when the estimated increase in loads of DIN or TSS was relatively low (i.e., <25%) to 1.0 (i.e., maximum importance) for catchments with very large increases in estimated loads of pollutants (>1,000% for DIN and >350% for TSS). Due to high variability in the proportional change in pollutant loads between catchments and pollutants, I assigned different intervals for CF categories for TSS and DIN, in both cases set based on a geometric classification.

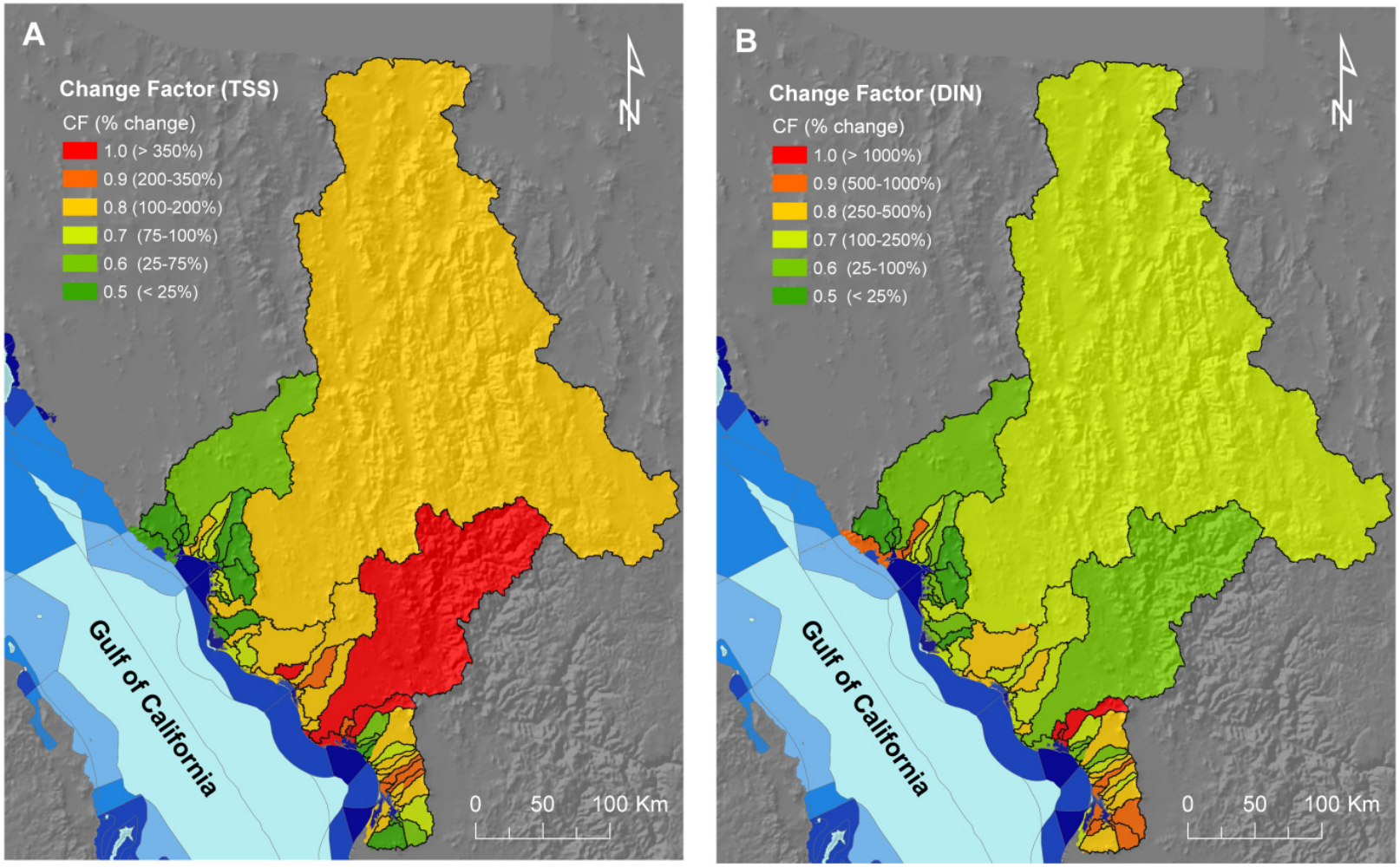

Figure 6.6. Change factors of the thirty-nine catchments allocated according to the estimated change in total suspended sediment ï TSS (A) and dissolved inorganic nitrogen ï DIN (B) loads from the ónaturalô to ócurrentô (2000) land use conditions.

Scenario 2: protection of natural vegetated areas

Water quality objective 2: maintain coastal-marine water quality through protection

The second water quality objective (WQ2) concerned maintenance of end-of-river water quality to prevent impacts of land-based pollution on coastal-marine ecosystems. This objective aimed to avoid as much additional loss of water quality as possible, by targeting the subcatchments with both the highest risk of transformation and the highest potential TSS loads if transformed. I arbitrarily set this objective as 30% of the potential contribution from subcatchments with remaining natural vegetation that could potentially contribute the most to the region-wide TSS load if converted into anthropic land uses. As for the previous water quality set of objectives (i.e., management of anthropic areas), I incorporated the link to marine areas by scaling the modelled maximum proportional contribution of TSS of each subcatchment (TSS_{max}, **Figure 6.4**) based on the marine priority (MP) and proportional change (CF) factors using the following function:

$$TSS_{scaled} = TSS_{max} \cdot MP \cdot CF_{TSS}$$

Biodiversity objective: conserve terrestrial vertebrates

The second objective regarding protection of natural vegetation was to protect natural areas to conserve terrestrial biodiversity, in this case vertebrate species (47 mammals, 80 birds, 43 reptiles, and 16 amphibians). Variable objectives were set for different species, ranging from 5% to 100% of their potential distributions. I used the objectives that the Mexican government determined for species based on their conservation and protection status following expert advice (CONABIO 2007b). I used expert-validated modelled potential distributions generated with the GARP modelling tool (Stockwell 1999) as raster grids (1 km resolution) for: amphibians and reptiles (Flores-Villela & Ochoa-Ochoa Unpublished), birds (Navarro & Peterson Unpublished), and mammals (Ceballos 2007).

Comparing selection frequencies for different objectives

The maps of land values were the ósummed solutionô (i.e., selection frequency) and óbest solutionô from Marxan analyses. The ósummed solutionô shows the percentage of the 100 solutions in which each subcatchment was selected. The óbest solutionô shows one possible configuration of subcatchments that achieves the defined objectives and spatial requirements for least cost (i.e., the most efficient solution of the 100 Marxan runs, see Ardron et al. 2008). I compared the ósummed solutionô maps from the three scenarios to identify the areas of coincidence (comparable frequencies and congruence between óbest solutionsô) and areas of divergence (different frequencies, indicating the need for tradeoffs between objectives). For the scenarios regarding protection of vegetation (i.e., terrestrial biodiversity and water quality for TSS erosion) I created a map depicting the differences by subtracting the two ósummed solutionô maps, as well as a map identifying the areas of coincidence in selection frequency, i.e., the number of times that each unit was selected in both scenarios.

Results

The results of my Marxan analyses showed important differences in the spatial distribution of coastal-marine and terrestrial land values, for both the management and conservation scenarios (**Figure 6.7**). Differences between ósummed solutionsô for achieving water quality objectives through management and protection were particularly striking (**Figure 6.7A, B**). However, these were expected because selections for management objectives targeted anthropic land uses for improving water quality (see **Figure 6.1, 3)** and selections for protection targeted naturally vegetated areas with high erosion and clearing potential (**Figure 6.4**). The Marxan results for the water quality objectives followed expected patterns. Marxan ósummed solutionô for the protection objective (**Figure 6.7B**) closely followed the spatial pattern of the targeted feature (i.e., higher selection frequencies correspond with subcatchments potentially contributing the most to the TSS load, **Figure 6.4**). Marxan ósummed solutionô for best-practice land uses was less clumped (**Figure 6.7A**), which can be partially related to the patchy distribution of anthropic land uses (**Figure 6.1**), but also to the divergent patterns of the different anthropic land uses targeted for TSS and DIN (**Figure 6.4**).

A comparison of the óbest solutionô outputs from Marxan for the two water quality objectives showed almost no overlap (only two subcatchments), indicating that subcatchments with native vegetation and high erosion potential (valuable for protection) and high current supply of TSS and DIN (valuable for management) were rare. The overall characteristics of the óbest solutionô outputs for the management and protection scenarios also differed notably. For instance, the óbest solutionô for the management scenario contained 134 subcatchments (8.1% of total number) and covered 8,600 km^2 (8.4% of the study area). Conversely, the óbest solutionô for the protection scenario contained only 67 subcatchments (4.1% of total number) and covered 5,880 km^2 (5.8% of the study area).

The ósummed solutionô for terrestrial vertebrates was also clustered, with higher selection frequencies concentrated in the upper regions of the Yaqui and Mayo catchments. There was some coincidence between the ósummed solutionô for managing water quality and for protecting vertebrate species, indicating the potential to achieve biodiversity objectives through the management of some of these areas. The number of units and total area of the óbest solutionô output for terrestrial vertebrates was much larger (i.e., 244 units, 14.8% of total number, and covering 21,611 km^2, 21.2% of the study area) than the óbest solutionô for the other two scenarios.

Figure 6.7. Priorities for catchment management and protection, depicting the relative value or priority of subcatchments for achieving two sets of marine objectives (improving or maintaining end-of-river water quality) and one set of objectives for conservation of terrestrial vertebrates. The value of planning units is represented by the ósummed solutionô maps of Marxan. Subcatchments selected more frequently in Marxan runs (darker colours) indicate their higher importance for achieving objectives. Also shown in the maps are Marxan óbest solutionsô for each set of objectives. A) Priorities for management to improve coastal-marine water quality by implementing best management practices in anthropic land uses (cropland, pasture and urban); B) Priorities to maintain water quality through protection of natural vegetation to minimise the increase in sediment loads delivered to coastal-marine areas; and C) Priorities for conservation of terrestrial vertebrate species.

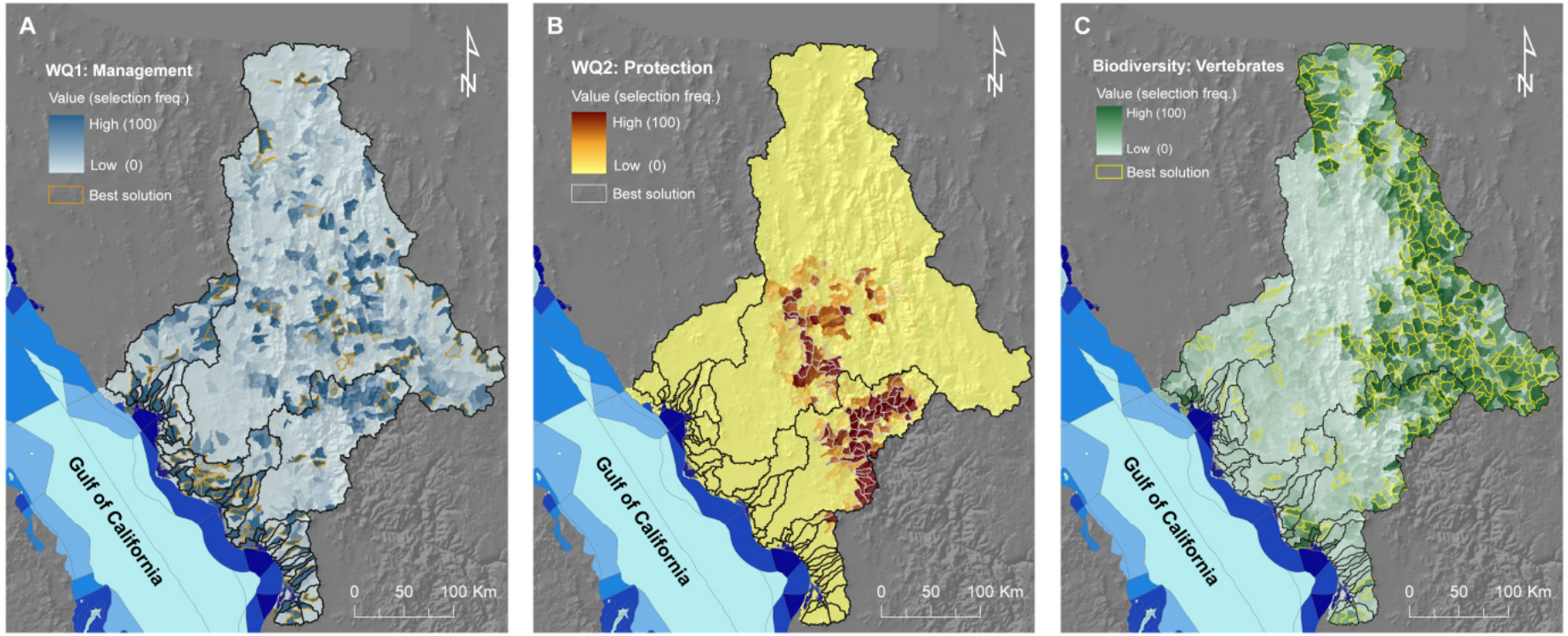

Figure 6.7. Priorities for catchment management and protection, depicting the relative value or priority of subcatchments for achieving two sets of marine objectives (improving or maintaining end-of-river water quality) and one set of objectives for conservation of terrestrial vertebrates

My comparative analysis of spatial similarities and differences highlighted major differences between the ósummed solutionô for water quality protection and for conservation of terrestrial biodiversity. A visual comparison of these two maps (**Figure 6.7B, C**) also revealed that very few subcatchments were selected as part of the óbest solutionô in both scenarios. Subtracting the two ósummed solutionô maps (**Figure 6.8A**) highlighted these differences. Achieving the objectives represented by these two scenarios would require an area almost equivalent to the sum of the two óbest solutionô maps, but there were some areas of coincidence as well (represented as the areas of coincidence between the ósummed solutionô maps: **Figure 6.8B**), raising the possibility of protecting some common areas to achieve both objectives.

Figure 6.8. Spatial similarities and differences between priority maps for the water quality protection and terrestrial biodiversity objectives. A) Differences between the two maps calculated by subtracting the two ósummed solutionô maps: red areas were preferentially selected for terrestrial biodiversity, while blue areas were selected more often for protection against erosion. Areas in yellow and light red or blue had comparable selection frequencies for the two sets of objectives (either low or high, but most of these areas depict areas of low selection frequencies). B) Areas of coincidence, measured as the number of times units were selected in both scenarios.

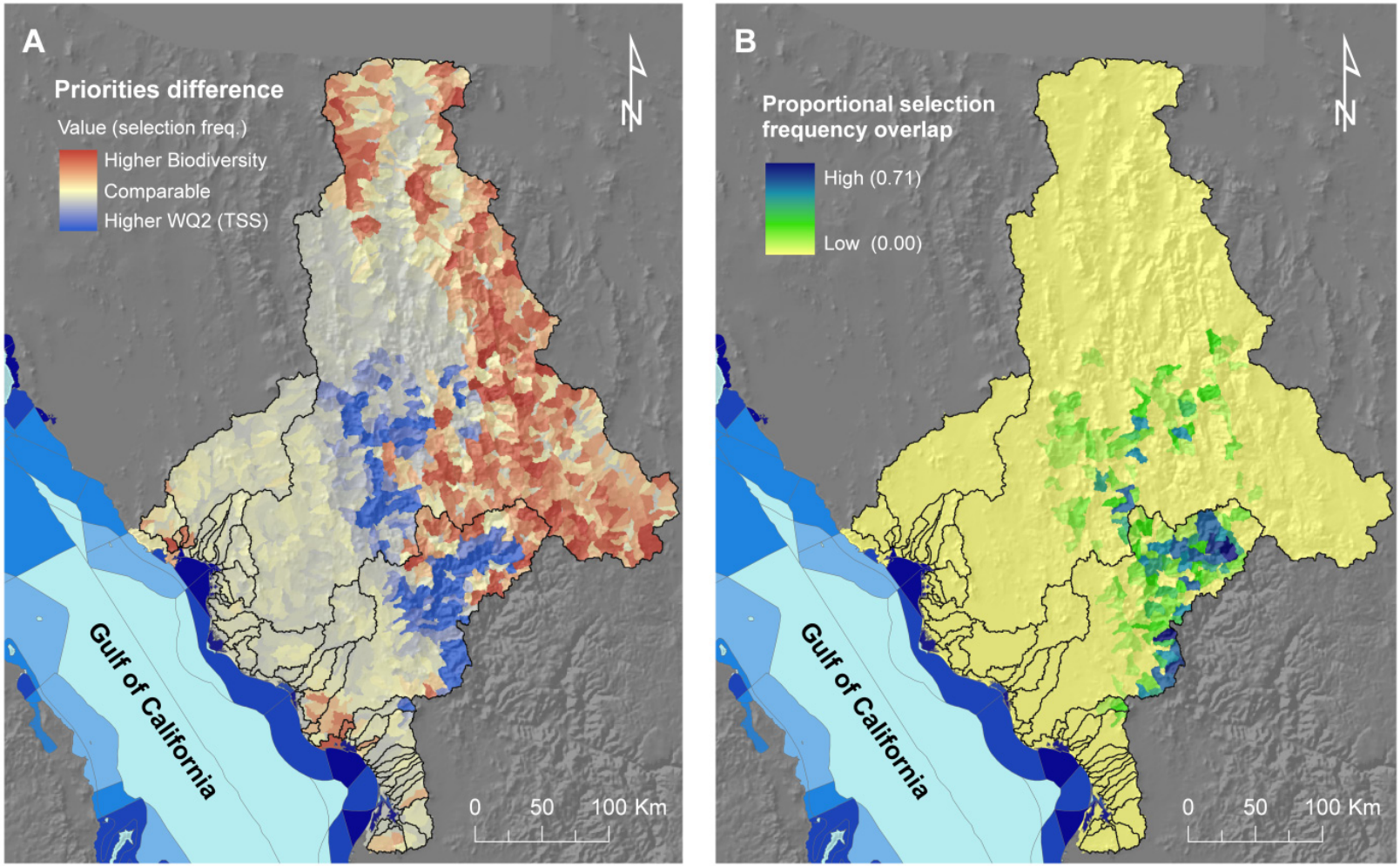

Figure 6.8. Spatial similarities and differences between priority maps for the water quality protection and terrestrial biodiversity objectives

Discussion

My findings highlight for the terrestrial realm what other studies have found for marine conservation priorities when land-based impacts were considered (Klein et al. 2010; Tallis et al. 2008). I found important differences in the spatial distribution of land values derived from terrestrial and marine objectives, just as Tallis et al. (2008) found differences in marine priorities when land-based threats were ignored or avoided. The minimal congruence between the priorities for conserving land vertebrates and to protect natural vegetation to maintain water quality illustrates the potential occurrence of competing objectives in integrated catchment management. Not surprisingly, similarity between the priorities for improving water quality through best-management practices and protection against potential erosion was also minimal. In this case, spatial differences are indicative of the targeted features and the types of actions to be implemented to achieve these objectives. The first scenario involves improving management in subcatchments dominated by anthropic land uses (e.g., through the implementation of agricultural and grazing best practices, Wilkinson & Brodie 2011) and the other involves the protection of natural vegetation to avoid clearing and hence erosion (Atherton et al. 2005). Despite the observed differences between the different scenarios, there are spatial options to manage areas simultaneously to achieve terrestrial biodiversity and water quality objectives. This is a promising result, and indicates that integrated land-sea planning can help to integrate and accommodate multiple objectives with benefits for both land and marine biodiversity, while reducing the overall cost of interventions.

While this study provides an important advance towards integrating land-sea conservation, it does not yet reach full integration and results should be considered preliminary. The study had two key limitations. First, I only provided a cursory link between end-of-river loads of pollutants and the marine environment (Halpern et al. 2008). The model of exposure to river plumes proposed by Devlin et al. (2012b), with the improvements proposed in Chapter 5, is a good alternative to be explored for the Gulf of California. For instance, adapting the model for the Gulf would necessitate differentiating chlorophyll-rich plume waters from periodic upwelling of nutrient-rich water resulting in colourful phytoplankton blooms (Beman et al. 2005) and developing a

new set of spectral signatures adapted to the region. Ideally, this exposure model should incorporate potential impact on affected marine ecosystems, which could be informed by expert opinion in the absence of spatial data regarding the effects of the modelled pollutants on particular ecosystems and species.

A second limitation is that I used pre-existing marine conservation priorities instead of targeting marine and terrestrial conservation areas simultaneously. This approach was necessary to accommodate previously set marine priorities resulting from an extensive consultative marine planning process. The marine spatial plan for the Gulf of California currently guides actions for conservation and management of coastal-marine areas in the region (SEMARNAT 2006). Fully integrated land-sea planning would ideally determine marine and terrestrial priorities simultaneously so that maximum benefits of conservation and management actions can be achieved for land and sea (and preferably for freshwater ecosystems too), and costs of interventions are minimised across realms. A next step to simultaneously minimise land-based threats and prioritise marine interventions may require adapting conservation planning tools/algorithms to explicitly incorporate connectivity between land and marine planning units (some options to adapt conservation planning tools to account for connectivity are provided by: Beger et al. 2010a; Beger et al. 2010b).

Conclusions

The need to integrate land and marine conservation planning has been pointed out by several studies (Beger et al. 2010a; Stoms et al. 2005), yet to date very little emphasis has been placed on examining terrestrial management for achieving marine conservation goals (Alvarez-Romero et al. 2011). I provide one of the first approaches to prioritising land conservation and management for marine conservation goals following a systematic conservation planning approach (but see Hazlitt et al. 2010; Klein et al. 2010). Previous studies show the potential for competing land values derived from different management objectives (e.g., to maintain the provision of different ecosystem services such as water retention, carbon storage and flood control: Chan et al. 2006), but my study is the first comparing priorities for land management for marine conservation and priorities for terrestrial conservation. Other studies have linked sources of land-

based pollution to affected marine areas using catchment and river plume models (Crist et al. 2009; Paris & Cherubin 2008), but none that I know of has integrated these modelling tools with land use change models to prioritise catchment protection. Identifying natural areas prone to erosion is the first step to identify the areas potentially contributing to sedimentation (Atherton et al. 2005), but considering land use change is essential to prioritise interventions in areas more likely to be cleared. My study proposes a method that integrates land use change, catchment and river plume models with conservation planning software to prioritise catchment management. Unlike Klein et al. (2010), who did their analysis at the resolution of large ecoregions, my approach is at a resolution more amenable to management actions (subcatchments). While the modelling required to achieve this kind of land-sea conservation integration is complex and time-consuming (Bohnet et al. 2011; Crist et al. 2009; Paris & Cherubin 2008), I show that it is possible and feasible, but also necessary to better guide integrated land-sea planning exercises.

Chapter 7. General discussion

In this chapter I summarise the key findings and how I addressed the objectives of my thesis, identified in the introduction and reiterated below. I also discuss how my thesis contributes to the scientific basis for integrating land and sea into planning processes.

Achievement of thesis goal and objectives

The overarching goal of my thesis was to improve understanding of fundamental elements in land-sea conservation planning, and investigate how these can be incorporated into the planning process. I achieved this goal through three research objectives:

1. Assess progress in land-sea planning theory and practice, and identify landmarks and challenges (Chapters 2-3);

2. Develop novel methods for incorporating cross-system threats by mapping and targeting land-based threats (Chapters 4-5); and

3. Explore coincidence between land values derived from terrestrial and marine conservation objectives (Chapter 6).

For Objective 1, I developed an operational framework to guide integrated land-sea conservation planning (Chapter 2). Key advances of this chapter included: a classification of land-sea processes, which also identifies potential planning strategies (design criteria, variable objectives, planning units, and movable areas) to incorporate these processes into planning. These strategies build on previous studies incorporating land-sea processes (e.g., Hazlitt et al. 2010; Lombard et al. 2007), and are placed in the context of an existing classification proposed by Beger et al. (Beger et al. 2010a). I also identified major cross-system threats and elements to consider (sources, affected realms, directionality of influence, stressors, and areas for intervention). Previous studies have identified necessary elements for incorporating influence of land activities on marine ecosystems (Stoms et al. 2005) and explored potential strategies (e.g., avoiding placing

conservation areas under the influence of land-based threats) to incorporate cross-system threats (Tallis et al. 2008). I use a decision tree to illustrate alternatives to address cross-system threats, other than avoiding them, which depend coastal management (Brodie et al. 2009; Kroon et al. 2012) and vulnerability of marine areas affected (Halpern et al. 2007).

I used the operational framework (Chapter 2) to assess methodological similarities/differences and identify planning gaps, using the Gulf of California as a case study (Chapter 3). Previous studies have explored the benefits of a systematic approach to marine planning for some regions (Fernandes et al. 2009; Gleason et al. 2010; Osmond et al. 2010), but my study is the first to investigate the complexities of multiple planning in the same region. The review of planning exercises indicated that the use of systematic conservation planning methods and tools has progressed slowly in the region, and major gaps include planning for land-sea connections and ecosystem services. These findings are in accordance with previous studies recognising the need to further develop methods to better understand and incorporate processes (Edwards et al. 2010; Grantham et al. 2011; Lagabrielle et al. 2011) and ecosystems services (Naidoo et al. 2008; Silvestri & Kershaw 2010). I also found some spatial convergence between the maps of priority areas (e.g., in nearshore areas, some habitats and ecoregions), but small areas of high coincidence. In addition to guiding MPA implementation, other benefits of planning included developing skills and knowledge, improving collaboration, and building trust. My study complements the findings of previous studies (Bottrill et al. In review), identifying benefits of planning other than direct protection. Based on expert opinion I conclude that multiple plans in the same region can result in some benefits (e.g., improved methods, better data and tools, basis for other plans) and difficulties (e.g., confusion, redundancy, inefficient use of resources).

To achieve Objective 2 (developing novel methods for incorporating cross-system threats), I explored methodological approaches to mapping sources of land-based pollutants and estimate end-of-river loads for integrated land-sea conservation planning (Chapter 4) and developed a novel approach to assess exposure of coastal-marine ecosystems to river-borne pollutants (Chapter 5). In the former, I compared two catchment models commonly used to target catchment management (N-SPECT and SedNet). Previous studies have examined nutrient enrichment of coastal waters in the

Gulf of California, but focused on irrigated agriculture (Ahrens et al. 2008; Riley et al. 2001), and local scales (Christensen et al. 2006). My study focused on the application of catchment models to incorporate cross-system threats (Tallis et al. 2008) into conservation planning, a recognised need in the region (Alvarez-Romero et al. In review; Lluch-Cota et al. 2007). My results indicated that both models are generally applicable under limited data circumstances, but SedNet requires additional datasets and parameters. The patterns of supply are similar and outputs of both models are adequate for prioritising long-term management in land-sea planning applications. However, the divergence between modelled and observed TSS loads limits the use of N-SPECT for plume & ecological modelling. Additional functionality in SedNet offers useful outputs for targeting catchment management; for TSS in particular, SedNetôs ability to include reservoir trapping is critical, considering their potential influence in end-of-river loads (Wilkinson et al. 2009).

For the development of a model of exposure to river plumes (Devlin et al. 2012b; Maughan & Brodie 2009), my results indicate that classification of true colour satellite images can be used to map plumes and to qualitatively assess exposure to river-borne pollutants. This approach should be considered complementary to remote sensing methods based on ocean colour products used to characterise surface water in plumes (Qin et al. 2007; Schroeder et al. 2012). The model is useful to study the spatial and temporal variation in exposure of coastal-marine ecosystems to riverine plumes. The observed inter-annual variation in exposure of habitats to pollutants stresses the need to incorporate the temporal component in exposure and risk models. This can be particularly important to better understand the impact of extreme weather events on vulnerable marine ecosystems and species (Devlin et al. 2012a; McKenzie & Unsworth 2011).

I achieved Objective 3 ï exploring coincidence between land values derived from terrestrial and marine conservation objectives ï by developing a method that integrates a catchment model, a land use change model, and a river plume model for land-sea conservation planning applications (Chapter 6). My results emphasise that integrating land use change in spatial prioritisation procedures is needed to inform management for terrestrial conservation (Laurance et al. 2002) and water quality (Burke & Sugg 2006). Integrating catchment and river plume models is thus critical to assessing effectiveness

of management actions to achieve water quality objectives. Notable differences in the distribution of land values for terrestrial biodiversity and coastal-marine conservation (end-of-river water quality) indicate the potential for conflict (i.e., competing objectives and the need to analyse trade-offs). The spatial incongruence between maps developed based on different land management objectives has been previously recognised (Chan et al. 2006), but this study is the first comparison of values derived from marine and terrestrial conservation objectives. However, there are spatial options to accommodate priorities to achieve terrestrial and marine objectives by managing land areas that also provide downstream benefits. The proposed method contributes to the body of literature investigating integration of tools to better inform catchment management and conservation planning (Bohnet et al. 2011; Crist et al. 2009)

Original contributions and findings

Each of my thesis chapters contributes to the advancement of conservation planning science theory and practice:

First, I developed an operational framework to guide land-sea conservation planning (Chapter 2). Novel aspects of the chapter include the development of a framework that builds on the general framework of conservation planning (Pressey & Bottrill 2009), but emphasises the elements that are particularly relevant to land-sea planning based on the importance of land-sea connections (Stoms et al. 2005). The framework identified three fundamental elements underlying the need for land-sea planning: cross-system threats, land-sea processes, and socioeconomic interactions across realms. The chapter also provides practical advice to conservation practitioners by identifying key elements important for planning ï determined based on expert opinion and case studies ï , as well as the methods and tools to facilitate integration.

Chapter 3 is a novel contribution to the literature because it is the first study that I am aware of that explores the peculiarities of having multiple planning exercises in the same region. I illustrate that plans differ because of their methods, data used, and planning extent. Informed by expert opinion, I confirm some of the findings from previous studies (Bottrill et al. In review) that indicated that planning has other benefits

(e.g., capacity building, improve collaboration, compile data and identify gaps) apart from informing siting of MPAs. The chapter also contributes to conservation practice by identifying key gaps and ways forward for refinement of existing plans in the Gulf of California.

My comparison of catchment models (Chapter 4) was specifically aimed at testing models that could be used at a regional scale to integrate land-based threats for marine conservation planning applications. My study advances conservation planning theory and practice by highlighting elements of catchment models most relevant for marine planning applications, providing a simple approach to comparing catchment models, and a relatively simple way to summarise and compare model outputs for use in prioritisation applications, which then can be incorporated into multiple objective analyses (Bohnet et al. 2011; Chan et al. 2006). The compilation and preparation of the data was very useful to identify data gaps and to direct future research priorities for the Gulf of California.

The model of exposure to land-based pollutants proposed in Chapter 5 is the first to use classification of true-colour imagery to identify and characterise river plumes, thereby providing way to link sources of land-based pollution and affected marine systems. The study also contributes to the understanding of long-term spatial and temporal variations in exposure of vulnerable marine ecosystems to plumes (and to specific pollutants). This information can help to explain some of the observed patterns of disturbance (e.g., decrease in seagrass and coral cover), and thus is the basis for developing impacts models to better inform marine planning (Devlin & Schaffelke 2009; Halpern et al. 2009).

In Chapter 6 I integrate for the first time land use change, catchment and river plume models to prioritise catchment management, aiming to improve protection of vulnerable marine areas. I illustrate a method to integrate these models into prioritisation analyses using conservation planning software. Previous studies compared maps of priority areas for maintaining the delivery of different ecosystem services (Chan et al. 2006) and priorities derived from terrestrial (forest biodiversity) and marine (terrestrial nesting habitat of a seabird) objectives (Hazlitt et al. 2010), but my study is the first to compare priorities for terrestrial biodiversity and for coastal-marine water quality. Identifying

these differences and the potential for competing objectives is a necessary step to explore the tradeoffs associated with management decisions.

An additional contribution to conservation planning is the undertaking of modelling exercises in the Gulf of California region under relatively limited data and resources. These circumstances are likely to occur in many other regions, in particular in developing countries. My efforts illustrate that a lot of analyses can be accomplished even with limited data and resources.

Potential practical applications

In conservation planning, facilitating conservation can be as important as making theoretical scientific contributions. My thesis promises to have several applied, practical contributions to conservation through the data and models I developed, and my engagement with practitioners throughout my thesis.

An important contribution of my thesis to conservation practice is the compilation, collation and correction of data for the Gulf of California. I organised and documented all datasets in a database. I plan to contribute these datasets to a permanent geographic information system repository that is under development for the Gulf of California. The inclusion of these datasets and outputs from my modelling analyses will be an important addition to this system, and will facilitate integration of my approaches into conservation practice.

The process of compiling information led to my engagement with governmental and non-government organisations involved in conservation planning initiatives, and I have been invited to develop or contribute to ongoing planning initiatives in the Gulf of California region. In particular, I have been invited to participate in a proposal to develop a plan for the creation of a network of marine no-take areas in the Midriff Islands region of the GoC. I am also engaged in an exercise to integrate the terrestrial, freshwater and marine priorities derived from the National Gap Analyses. Finally, there is interest from the Protected Areas Commission to undertake a series of pilot land-sea planning exercises for several coastal catchments. For the two latter initiatives, there is

opportunity to make use of the operational framework, tools and datasets generated for my thesis.

For the exposure model developed for the Great Barrier Reef, there is already interest from researchers and managers (e.g., from the GBR Marine Park Authority) in using the exposure maps. A primary goal of this modelling exercise was to develop a tool that could complement the activities under the Marine Monitoring Programme of the Marine Park Authority. In combination with the ongoing *in situ* monitoring programme and other monitoring strategies, the exposure model could become a tool to monitor water quality of coastal-marine ecosystems of the GBR.

Limitations, opportunities for improvement and further research

Every research project has limitations, and mine was no exception. Throughout my research project I recognised several limitations, including those of scope and methods, and encountered some problems that highlighted additional shortcomings. Overcoming some of these limitations will require the compilation of new data, expanding the study area, ground-truthing, and validating some outputs of the models. It may also require adapting or adjusting the methodology, or even exploring different approaches. To some extent, these limitations can constrain the interpretation of my results or the applicability of my outputs. Despite the recognised opportunities for improvement, my research dealt with these limitations in the best way possible given the time and resource constraints.

The operational framework for integrated land-sea planning developed in Chapter 2 provides a comprehensive reference for guiding planning exercises, and emphasises the elements that merit further research. Nonetheless, the rapidly advancing science of conservation planning and, in particular, the increasing number of marine and land-sea applications will likely require updating the framework. An avenue for further research in this regard is the revision of the framework with an emphasis on documenting the interoperability of modelling tools to incorporate land-sea connections. Some aspects may require major adjustments, such as socioeconomic considerations, planning for processes and implementation, because they are developing very rapidly. To keep the

framework up-to-date, it will be very important to continue updating the database of case studies developed for Chapter 2.

The review and spatial analysis of marine planning exercises of the Gulf of California (Chapter 3) provides a good summary of the state of marine planning in the region. However, the poor documentation or absence of spatial datasets used in some of the plans limited my comparison. On the other hand, the diversity in approaches and limited documentation of some plans required intensive work with planners to document the methods followed. The poor documentation of plan implementation prevented me from undertaking a systematic retrospective analysis of the implementation of spatial actions associated with specific plans, as I had initially hoped to do. Hence, I relied mostly on anecdotic information provided by planners and experts, which was insufficient to assess uptake of planning outputs systematically. Access to information regarding planning processes at a local scale (e.g., rezoning of individual MPAs) was in general very limited, thus it would be desirable to incorporate these exercises in future iterations in the database of plans in the Gulf of California. Further research regarding marine planning in the Gulf of California may include the development of a framework to follow up on the implementation of existing plans, as well as to integrate existing priority maps in future prioritisation exercises (e.g., rezoning, refining boundaries of conservation areas, local-scale plans, planning within marine priority areas). In relation to land-sea integration (a recognised gap in marine planning for the region), the methods proposed in Chapter 6 could guide further prioritisation analyses.

Catchment modelling in a region with relatively limited data means that data gaps constrain the results. My study (Chapter 4) should thus be interpreted as a comparison of the two models under the limitations imposed by available data and given the characteristics of the study area, not as a generic comparison of the models under all circumstances. Higher quality inputs would likely improve the outputs of both models. In particular, to improve the accuracy of catchment models, research and monitoring activities in the study area should improve soil data, map and study the contribution of gullies to TSS, characterise local erosion rates, and monitor catchment-scale and long-term nutrient loads associated with rainfall runoff. A lesson I learned during this chapter was the significant amount of time it takes to generate these models ï a possible reason why so little land-sea planning has been done. What had been intended as just one

component of my attempt to integrate land-sea planning turned into a 2-year undertaking, the most time-consuming part of my thesis.

Chapter 5 contributes to understanding the spatial-temporal dynamics of plumes in the Great Barrier Reef (GBR), and offers a method that can improve models of plume exposure developed for the GBR and elsewhere. However, some limitations are inherent in my plume modelling approach, especially the potential confounding influence of other surface water masses that resemble plume waters (e.g., associated with resuspension or upwelling) and the focus on surface exposure that thereby ignores the vertical extent of the plume or the mixing in the water column. More work is needed to further understand how the pollutants carried in the flood plume become incorporated in the water surrounding the benthic habitats below. Some of these limitations could be addressed by using hydrodynamic models to predict the dispersal and mixing of plumes and pollutants, but such models are not available in most regions, or if available, are not at resolutions conducive to plume modelling. Another potential improvement is the modelling of exposure at finer temporal scales. If weekly load data are available (e.g., linked to a catchment model such as Source Catchments: Carroll et al. 2012), the model can be adjusted to produce exposure models that represent variability at weekly time-steps. Further investigations that can improve the model include: validation of the selected TSS/DIN proxies against *in situ* measurements, test of other proxies, exploration of the effects of using an *ad hoc* RGB stretch when producing true-colour images, and adjustment of spectral signatures used in the supervised classification to account for potential seasonal and/or inter-annual variations. One disappointment was that I hoped the approach would be generically applicable (i.e., that the spectral signatures of plumes would be transferable to other regions) so that I could use it for the Gulf of California. My attempt to do this, however, uncovered the above-mentioned issues of other water masses, which are much more prevalent (especially upwelling) in the Gulf of California.

The method developed in Chapter 6 represents the culmination of my attempt to do land-sea planning, and it is an important advance towards integrating land-sea conservation. However, it does not yet reach full integration ï I did not prioritise marine regions at the same time as terrestrial ones. Simultaneous prioritisation would be preferable because it would facilitate the creation of integrated solutions that maximise

the benefits for conservation and management actions for land and marine objectives. Also, the link between land-based sources of pollution and the marine environment was not as sophisticated as I had hoped. Unfortunately, as mentioned above, implementing the suggested exposure model in the Gulf of California did not work out because it would have required significant adjustments.

Future work

The limitations of my thesis represent opportunities for future research, and I will be addressing several of these in future research work. In particular, I will work on an updated framework for integrated catchment planning, which will explicitly include sustainable development objectives, and describe with more detail interoperability between modelling tools. Part of this research will include adaptation of existing conservation planning software (e.g., to incorporate multiple objectives, costs and links between planning units functionally connected through streams) to automate some of the prioritisation analyses undertaken in my study, and to generate ôoptimalô solutions based on multiple objectives. This research will be undertaken in catchments of Northern Australia and probably also include pilot studies in the Pacific Coast of Mexico and within the Gulf of California. As previously acknowledged, collaboration with stakeholders was a priority during my research to ensure uptake and outreach of my findings. For this reason the work in Australia places a big emphasis in developing the research in close collaboration with natural resources managers and communities from the onset of the project.

My established collaborations with the Catchment to Reef group of the Centre for Tropical Waters Research (TropWater) at James Cook University will lead to further refinement of the exposure model. Improvements will include the automation of some of the analyses, as well as the creation of a reporting tool that can be used to inform monitoring of water quality in coastal-marine waters. These tools can then be transferred to other research groups and be adapted for different regions. Further research will also include validation with in situ data, and analyses to explore the potential relationship between estimated exposure and trends in the condition of vulnerable coastal-marine ecosystems (e.g., coral reefs and sea grass beds). This will

allow to develop better indicators and risk assessment, as well as to improve catchment management. Mapping of plumes (based on true colour) associated with dredging activities is another area of research that could be explored.

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Appendices

Supplementary Material C2: Case studies summarising selected elements relevant to land-sea integration in conservation planning

Planning exercise	*Goals*	*Land-sea conservation objectives*	*Land-sea processes*	*Cross-system threats*	*Integration approach*	*References*
Alaska Cook Inlet (United States)	Assess ecoregion biodiversity and identify areas of biological significance in terrestrial, freshwater, and marine realms.	Anadromous streams for fish were explicitly targeted. Salmonids were selected because they are ecological drivers of aquatic systems in south central Alaska. Species aggregations, such as foraging areas for migratory birds or feeding concentrations of bears in rivers to feed on salmon were recognised as important features to be protected.	Some watersheds were selected as integral units to protect full sets of runs of anadromous species because of their importance to upstream nutrient subsidies. Post-design and connectivity criteria helped develop these selections. Key conservation areas included estuaries with resting sites for salmon during their upward spawning migration and for young on their seaward movement to marine waters.	Dams and storm water discharges and other threats were incorporated into the cost suitability index. In the selection of the portfolio of aquatic areas, the health of aquatic systems, dependent on the condition of lands in the watershed, was also taken into account (i.e., downstream impacts).	SITES software (DSS) was run for terrestrial and coastal/marine systems simultaneously. The DSS output was then overlaid with polygons defined by experts to construct a draft portfolio. This portfolio was further refined in an expert workshop, where priority was given to areas containing multiple types of systems (e.g., terrestrial, aquatic, and coastal).	(TNC, 2003)
Alaska Peninsula and Bristol Bay (United States)	Assess the biodiversity of the Alaska Peninsula and Bristol Bay Basin ecoregions and identify areas of biological significance, including freshwater and marine environments.	Diadromous species: five species of salmon, steelhead trout, rainbow smelt, and dolly varden. These species transport sea-derived nutrients to freshwater systems and are important prey species to terrestrial and marine predators. They also occupy many habitats during their life cycle. Shorebirds occupy intertidal habitats and wetlands. Crabs rely on nearshore and coastal habitats.	Spawning migrations: large rivers terminate in broad estuarine areas supporting some of the largest salmon runs and fisheries in the world (e.g., greatest runs of sockeye salmon). Species aggregations included feeding areas (e.g., shorebirds and waterbirds congregate in the same food-rich lagoons during their spring and fall migration or in protected, ice-free estuaries as wintering areas).	Cross-system threats incorporated into the assessment included dams. Sites with higher costs were avoided in the portfolio unless the conservation features in such areas did not occur anywhere else in the ecoregions.	Portfolio was manually delineated by experts on the basis of other conservation plans and spatial data for fine-filter conservation features, including important areas for diadromous fish (some include whole-stream systems). Priority was given to areas that contained multiple types of coarse-scale systems (e.g., terrestrial, aquatic, and coastal); many priority areas incorporated whole catchments or subcatchments; areas chosen primarily for the marine environments included approximately a 5-km wide strip of shoreline.	(TNC, 2004)
Alaska Yukon Arctic (United States)	Integrate the best available information about the ecology of the region to identify lands and waters most necessary for maintenance of the ecoregionôs biodiversity.	Salmon included as conservation features.	The freshwater ecosystems model (used as surrogate for aquatic biodiversity) considered whether stream systems were connected with coastal areas. This was used as an indicator of potential presence of anadromous and amphidromous fish as well as estuaries at river mouths. It was recognised that a connection to marine waters affects the composition of biodiversity through upstream transfer of marine nutrients.	A single cost suitability index was constructed for terrestrial and marine habitats, and another for freshwater, but using the same data. Watersheds were nested, and therefore costs for freshwater were based on land management of the watershed and upstream watersheds.	SITES was run for all systems simultaneously. Watersheds up/downstream of each other were assigned low boundary lengths to favor their mutual selection; both best run and summed solutions were used, and these were further modified with expert advice on missed areas and unique sites (e.g., Chinook salmon spawning and migrating streams). Experts participated through data input, validation, and design during workshops.	(TNC, 2005)
Bering Sea (Russia, United States, international waters)	Identify the key biological features and ecological processes that define the Sea and contribute to its unique richness and productivity; develop a guide for biodiversity conservation efforts, but recognise the importance of communities and economies that live and work in the ecoregion.	Species that use land and sea habitats (e.g., shorebirds, waterfowl, seabirds), coastal habitats important for breeding colonies of marine mammals, migratory stopovers and staging areas were also considered. Important/sensitive habitats included eelgrass beds, coastal lagoons, wetland complexes, and estuaries. Diadromous species: five species of salmon. The run of sockeye salmon in Bristol Bay is the largest in the world, and Kamchatkaôs brown bear population, Steller sea eagles, and other wildlife are intricately tied to the Bering Sea by salmon.	Species, natural communities, and processes drove the final network of areas and associated strategies; consideration was given to habitat types and ecological processes needed for sustaining biodiversity of these areas, some of which are coupled to land-sea processes (e.g., concentration of squid along the shelf edge attracts salmon, which input nutrients to terrestrial and freshwater systems). Processes considered included those that take place at the ice edge or influence tidal and thus vegetation and forage patterns for coastal species, which could be altered by climate change and sea level rise.	Two expert-identified cross-system threats were included: point source pollution from coastal cities causing poor water quality and sedimentation, and chemicals from mining causing direct harm to species. Threats were ranked according to level of degradation, scope/scale of the threat (currently or potentially impacted), duration (persistence of threat), and urgency of action.	Experts advised on four taxonomic groups (birds, fish, invertebrates, and mammals) and also by regional expertise. Experts selected areas for major wildlife taxa based on five criteria: distinct or unique species assemblages, habitat or habitat complexes, richness or endemism, outstanding abundances or aggregations, and critical areas for sustaining important ecological phenomena. Maps from each of the four taxonomic groups were then combined to identify areas of concentration and importance common among all groups; consideration was given to ecological processes.	(Banks *et al.*, 1999)
Chesapeake Bay (United States)	Identify high-quality occurrences of defined conservation features to ensure their viability into the future. To achieve the goal of capturing all native biodiversity, the portfolio aimed to include terrestrial, freshwater, coastal, estuarine, and marine species and habitats. In particular, the objective of the freshwater analysis was to identify the most intact and functional stream networks and aquatic lake/pond ecosystems.	Estuarine, coastal, and marine species and habitat types common in and significant for ecosystem functioning of tidal waters and adjacent coastal uplands in the Chesapeake Bay and Atlantic coastal bays. Among the marine species, experts selected those whose life history includes activity in multiple ecosystems and which therefore provide indirect assessments of the state of those systems and the connectivity between them (e.g., anadromous fish, waterbirds, and waterfowl). Most of the primary targeted features are species that occur in, or are associated with, aquatic, wetland, or shoreline habitats. Tidal marshes, swamps, and other wetlands were combined for analysis into one habitat type rather than separating fresh and saltwater systems.	Indirect, not explicitly incorporated or targeted. Many of the conservation features are indicator species whose presence and abundance reveal good water quality, intact and functional ecological processes, and appropriate trophic structure.	A condition analysis was developed on the basis of the status and trends of water quality parameters (i.e., data on total suspended solids, dissolved inorganic nitrogen and phosphorus, chlorophyll-a, bottom/surface dissolved oxygen from a monitoring program) to assess status/impact of coastal habitats. The analysis could not be used to inform conservation areas because monitoring units were too large and comparable data were not available for the Atlantic coast.	Experts used design criteria: All else equal, an area encompassing a good diversity of conservation features and with high water quality would be chosen for the portfolio over an equally diverse area with poor water quality; however, the water quality condition analysis was not used because of data issues. Some marine sites are coincident with terrestrial sites that may decrease land-based threats and increase connectivity, e.g., largely forested to help control the amount of sediments and nutrients washed into the creek and adjacent river, and its upstream portions provided historical spawning grounds for anadromous fish species.	(TNC, 2002)

Coral Triangle	Maximise threat reduction to coral reefs across sixteen ecoregions through investment in a suite of conservation actions on the land and sea.	Identify if land or sea conservation actions deliver the greatest return on investment for coral reefs. In addition, identify where the greatest return on investment can be achieved.	Not considered.	Used mapped threats to coral reefs that could be mitigated with local-scale conservation action: 1. nutrient run-off from fertilisers; 2. organic pollution run-off from pesticides; and 3. artisanal fishing and five types of commercial fishing.	Mathematically formulated the resource allocation problem and determined the rate of return (i.e., reduction of threats to coral reefs) on investment (i.e., cost of reducing threats) of each conservation action (e.g., marine or terrestrial conservation) in each ecoregion. This way it can be determined if land or sea conservation delivers the greatest benefit for coral reefs.	(Klein *et al.*, 2010)
Equatorial Pacific (Ecuador, Peru)	Independent goals for terrestrial, freshwater and marine realms. Terrestrial: identify priority conservation sites within the terrestrial ecosystems. Freshwater: identify priority conservation sites, including watersheds and other hydrological systems, for adequate management of freshwater resources in the region. Marine: identify priority conservation sites within the marine ecosystems of the Equatorial Pacific region.	The terrestrial and freshwater teams developed conservation objectives at ecosystem and species levels, whereas the marine team organised objectives at ecosystem, natural community, and species levels. Features included estuarine fish species, marine intertidal (subdivided into estuarine and marine) and subtidal ecological systems, and animal communities (e.g., sea bird nesting colonies, migratory shorebirds).	Not explicitly incorporated or targeted. Incidentally included when targeting mangroves and recognised as an important component of estuarine systems and critical to maintain ecological processes between marine, freshwater, and terrestrial systems.	Models of cross-system threats were restricted to buffering cities to map the potential impact of coastal cities (proxy for domestic discharges) on the basis of population sizes. Accumulation in estuaries and bays due to limited circulation was also considered to modify the urban impact (cost values). Industrial pollution was also modelled, including mapping point locations of industrial facilities within 50 km of the coastline, classifying them based on the types of pollutants they produce, and projecting their discharges to points on the coast on the basis of their position within the watershed and in relation to rivers. Buffers at river mouths were also created for the different industry types.	Concurrent assessment of terrestrial, freshwater, and marine portfolios including a viability evaluation based on threats to species or ecosystems, connectivity within populations, and integrity of the main surrounding ecological processes.	(Ter§n *et al.*, 2004; Terneus *et al.*, 2004; Tirira *et al.*, 2004)
Fiji watersheds (Fiji Islands)	Identify watersheds important to protect and restore forest cover in order to protect healthy ecosystems and fisheries on high-conservation-value reefs. These include watersheds adjacent to reefs identified as highest priority by Fijiôs National Biodiversity Strategy and Action Plan. They are either well-forested and require adequate protection or are largely modified but need restoration.	Forests and priority coral reefs	Not considered	The Relative Erosion Prediction (REP) index was calculated for each catchment based on slope, soil erodibility, rainfall (total and seasonal), and current land cover to assess catchments' sediment and nutrient contributions to rivers and coastal ecosystems. A Watershed Development Index (WDI) was also calculated to assess the degree to which catchments were impacted by infrastructure. A composite threat index combined REP and WDI for each catchment.	No DSS or tools used to integrate realms. Simple overlay of critical watersheds with high conservation value reefs; both independent exercises.	(Atherton *et al.*, 2005)
Florida marine-estuarine (United States)	Provide resource managers, marine scientists, conservation practitioners, and other stakeholders with a tool to aid in the identification of a suite of areas that could serve as focal points for state-wide marine and estuarine resource management and conservation.	Many targeted species were characteristic of land-sea interface habitats, including crocodiles, estuarine turtles, anadromous fish (e.g., sturgeons, sawfish), and waterbirds. Interface target habitats, such as mangrove swamps and coastal tidal rivers/streams, also contributed to identification of linking elements.	Not considered	Land-based cross-system threats included in the socioeconomic suitability index were population and road density as well as major National Pollutant Discharge Elimination System (NPDES) permitted point source discharges. Scoring was developed and refined by expert review.	Marxan results were reviewed in expert workshops and utilised the collective advice of reviewers to develop a preferred portfolio. Determination of the most appropriate conservation actions to take at any particular site was left to a subsequent, more detailed site planning process	(Geselbracht *et al.*, 2005; Geselbracht *et al.*, 2009)
Florida Northwest Coast (United States)	Jointly meet objectives for coastal hazard mitigation and biodiversity. The study was developed as part of the state-wide marine-estuarine site assessment for Florida.	Included ecosystems and species that use/cover multiple realms: salt marsh, submerged aquatic vegetation (e.g., seagrass and tidal freshwater grasses), tidal flats, beach/surf zone, coastal tidal rivers or streams, bivalve reefs (oyster reefs), and inlets; Florida manatee. State-wide, features included species aggregations such as breeding concentrations and important nursery areas.	Indirectly included by focusing on the natural protection function that wetlands have on the coasts. The function of wetlands to mitigate coastal hazards to human communities (ecosystem service) was directly targeted. Targeting coastal wetlands function incidentally included other ecological processes supported by these ecosystems.	Hazard analysis: Areas at risk from storm surges were mapped using output from NOAA's Hurricane Storm Surge Inundation Model, which estimates storm surge heights and winds resulting from historical, hypothetical, or predicted hurricanes by taking into account pressure, size, forward speed, track, and winds. Graphical output displays storm surge heights for a particular area.	Two approaches. 1. Overlay: Coastal hazards (risk and vulnerability of human communities) were overlaid with data on biodiversity (coastal wetlands) through GIS to identify wetlands closest (within 5 miles) to human communities most at risk and vulnerable to coastal hazards. Conservation of biodiversity was a secondary objective. 2. Marxan: Wetlands identified in the overlay were targeted, along with other biodiversity features, to identify a set of natural areas that would most efficiently meet objectives for conserving biodiversity and mitigating coastal hazards simultaneously	(TNC, 2007)

Great Barrier Reef Marine Park (Australia)	Establish a new network of no-take areas to protect representative examples of the entire range of habitats, plants, and animals in the Marine Park while minimising impacts on existing uses.	Some habitats and species covered multiple realms: shallow water seagrass, dugong, islands. Ensure representation of reefs within the ñnaturalò diversity of water quality regimes (reef and non-reef areas in each of four flood frequency classes).	Two criteria for identifying zones were relevant to land-sea processes: integration with coastal habitat (e.g., mangroves, wetlands) and adjacency to national parks and other land/coastal conservation areas (e.g., lowland tropical rainforest, important seabird areas). This provided higher ecosystem integrity and maintained connectivity between land-sea habitats when possible.	Principles relevant to cross-system threats included: 1. Socioeconomic: Recognition of management and tenure (to maximise placement of no-take areas in locations that complement and include present and future management plans and tenure arrangements), and 2. Biophysical: Maximise natural integrity, considering adjacent uses in the sea and on land in determining no-take areas. All else being equal, areas of greater natural integrity (or adjacent to land of greater natural integrity) were included in new no-take areas.	Biophysical operational principles promoted the selection and location of no-take areas in accordance with land-sea integration criteria. Included consideration of sea and adjacent land uses in determining no-take areas. Two major aspects related to land-sea integration were: oceanographic diversity in water quality to ensure representation of reefs within the natural diversity of water quality regimes, and location of no-take areas adjacent to coastal and estuarine habitats (e.g., mangroves, wetlands) and land-protected areas rather than adjacent to suburbs to protect land-sea linkages and minimise impacts of land-based threats.	(Day, 2002; Day *et al.*, 2003; Fernandes *et al.*, 2005; GBRMPA, 2005; Fernandes *et al.*, 2009)
Gulf of California (Mexico)	Identify, through a highly participative process, the critical areas for biodiversity conservation and determine the relative level of anthropogenic pressure on natural resources as well as the potential for social conflict derived from conservation actions throughout the Gulf of California region.	Some targeted species, habitats, and ecological processes incorporate land-sea linkages, including marine turtles, seagrasses, mangroves, wetlands, and seabirds.	Experts delineated critical areas for conservation of biodiversity and important ecological and physical processes. Some of these processes occur in the land-sea interface, and some included coastal stream systems associated to coastal/marine features of interest. Priority areas were reviewed on the basis of their relationship to other areas, including land-sea integration.	A spatial index of anthropogenic pressure on biodiversity was developed. For land-based threats to marine systems, the index included point sources of pollution, which were mapped with buffers. The final anthropogenic pressure map covers terrestrial and marine realms, but the index was not used to inform the configuration of priority conservation areas. It was included as an independent analysis and overlapped with priority sites to find areas to target for conservation actions and to identify areas of potential conflict.	Areas were defined by experts supported by GIS software (WWF-CECARENA DataMap2) and selected by broad consensus through a highly participative process. Biologically important areas for the taxonomic groups and areas where important ecological and physical processes take place were overlapped to define integrated important biological areas. Design criteria included representing important ecosystems and habitat for multiple species (e.g., wetlands) and inclusion of important ecological processes, including those occurring in the land-sea interface.	(Enr²quez-Andrade *et al.*, 2005)
Jervis Bay (Australia)	Determine the range of practical options for conservation of marine and terrestrial environments (including natural and cultural values) of the Jervis Bay region. The strategic aims included an integrated approach to the management of marine and terrestrial systems.	Some targeted species, habitats, and processes incorporate land-sea linkages, including seagrass, some rare or threatened species, wader areas, seabird habitat, plants of special significance, subtidal sandy sediment fauna, and wildlife corridors. Seagrass communities were used as a surrogate for pollution.	One of the strategic aims ranked by experts in the selection of preferred conservation-management options was interrelationships to recognise and protect key processes rather than just specific landscape/ecosystem features.	Point/non-point sources of pollution from upstream development (i.e., rural and urban development, forestry) were identified as potential threats to marine habitats. Pollution of waterways was considered an important threat to the integrity of wetlands, salt marsh, mangrove, and seagrass communities (important habitats to the fishing industry and tourism). No explanation of the spatial incorporation of these threats is included in the report.	The E-RMS GIS software (NSW National Parks and Wildlife Service) was used to integrate and analyse data. One of the strategic aims was the consideration of integrated management. Priority was given to delineation of core conservation areas and their interconnecting elements (e.g., aquatic resources are to be managed in a complementary fashion at both the catchment and offshore marine area scales). The preferred management alternative included approximately 65% of the terrestrial part of the study area and the entire marine area, for which a zoning scheme was developed.	(Dutton *et al.*, 1994; Cho, 1998)
Kimbe Bay (Papua New Guinea)	Design and implement a marine protected area network to conserve marine biodiversity and natural resources in perpetuity and address local marine resource management needs.	Some targeted species, habitats, and processes included land-sea linkages, such as interface habitats (e.g., seagrass beds, mangrove forests, and estuarine breeding and nesting sites for seabirds) and key species (e.g., sea turtles, dugong, and seabirds).	Not explicitly incorporated or targeted. Incidentally captured by mangroves, which were targeted in three groups. One of the groups was associated with streams and bays as well as a large estuarine complex, emphasising the land-sea connection and the importance of interfaces as functional units. No specific conservation objectives or treatment was given to increase their selection.	Major land-based threats in the coastal catchments were mapped (palm oil plantations and logging). These threats are located mostly within the coastal plains, not far from upper catchments or mountains. Higher costs were assigned to areas adjacent to major towns and large rivers with industry by building a buffer in the mouth of rivers, major towns, and ports.	Marxan was used to develop alternative MPA design options. General design principles included potential for effective management (e.g., sea and land use, proximity to threats). Marine areas adjacent to unmapped terrestrial threats (e.g., large-scale agriculture and logging) were considered and if needed modified manually to follow design criteria. Areas with changing land use patterns were avoided when possible (e.g., predicted expansion of agriculture and village-based use).	(Green *et al.*, 2009)
Marshall Islands (Pacific Islands)	Maintain or improve biodiversity, health, productivity, and integrity of atoll ecosystems; sustain artisanal subsistence use of resources; and protect and preserve areas of significant natural and cultural heritage. Also, develop principles, process, and guidelines for the design, establishment, and management of conservation areas at local scales (i.e., no specific sites for conservation areas were identified during the first planning stage).	Some targeted species, habitats, and processes included land-sea linkages, such as seabirds, mangroves, sea turtle nesting sites, seagrass meadows, and land crab (an important land-sea connection because of its need to access marine waters to breed).	The proposed design principles (to be implemented at the atoll-scale level of planning) included land-sea adjacency of conservation areas. Local planning exercises should consider land-reef interactions and ensure that some managed areas on reefs and land are next to each other.	Design principles (atoll-scale level) included considerations for threats. Sites were given higher priority if they have low threats. Highly threatened sites can be selected only if they host critically threatened or unique species. This includes avoiding conservation areas/no-take reserves next to likely sources of land-based pollution and aquaculture sites.	Relevant land-sea design principles developed in the study included: avoid conservation sites with high threats (e.g., adjacent to pollution sources) and place marine and terrestrial areas adjacent to each other whenever possible to ensure natural interactions.	(RNPT, 2008; Baker *et al.*, 2011)

Mesoamerican Caribbean Reef (Central America)	Identify a network of conservation priority sites that will represent the regionôs biodiversity while contributing to the maintenance and resilience of ecological processes.	Some targeted species, habitats, and processes incorporate land-sea linkages: American crocodile, sea turtle, seagrass, mangroves, and ñisland-terrestrialò endemic vertebrate species and subspecies. Critical manatee habitat is found in several of the rivers, lagoons, seagrass meadows, and cays. Crocodiles are found along the regionôs rivers, coastal lagoons, islands, and cays.	Important biodiversity features considered included a diversity of interrelated habitats (e.g., seagrasses, mangroves, estuaries). The presence of seagrass, mangrove, and coral reef habitats in close association to one another was considered especially valuable because they provide a natural energy flow between systems and serve as corridors for organisms in transit and developing phases. Important seascape and integrity features used to evaluate persistence value included linkages, proximity, and connectivity to adjacent intact ecosystems.	Major catchments were classified based on their relative discharge as an indicator of potential impact associated with activities such as agriculture and urbanization. Cross-system threats were also incorporated into a general spatial assessment of threats that covered terrestrial, coastal, and marine habitats. The assessment included coastal development (changes in the hydro-geomorphological functioning of watersheds) and declining water quality (sedimentation, nitrification, and chemical contamination of estuaries, bays, wetlands, reefs, and seagrass beds). Experts evaluated the likelihood that threats would result in a loss, degradation, or alteration of terrestrial or marine habitats or processes.	Experts drew boundaries (polygons) of important areas. When identifying priority areas for focal taxa/guilds, criteria specific to coral reefs included proximity to nursery areas and ñswathò potential (i.e., extent to which an area contains or is linked to a variety of interconnected habitats and environmental regimes). The swath was envisioned as a full ñridge to reefò gradient fostering the areasô potential for holistic conservation integrating terrestrial, coastal, and marine ecosystems. An analysis of persistence of biodiversity areas included seascape and integrity features such as linkages, proximity, and connectivity to adjacent intact ecosystems.	(Kramer and Kramer, 2002)
Mesoamerica's Marine Ecoregional Assessment (Central America)	Identify priority sites for marine biodiversity conservation and develop a gap analysis for the region.	Some targeted species, habitats, and processes incorporated land-sea linkages, including estuaries and mangroves, seabird nesting colonies, migratory priority sites, and sea turtle nesting sites.	Not considered	Pollution discharges in river mouths that originate in the catchments were modelled on the basis of pollutant levels in river systems, population density in catchments, and catchmentsôindustrial development levels. Results were incorporated into a dispersal (logarithmic function) model to estimate spatial distribution (up to 15 km from coast). Coastal development was also modelled on the basis of major populations and infrastructure within 1 km of the coastline and mapped with decreasing impacts with distance from coast (500 m range).	Marxan was run for the whole marine region (including coastal and offshore areas), but analysis of areas >200 m depth were eliminated owing to the lack of data and resulting biased results. Land-sea integration was focused mostly on modelling and incorporating land-based threats.	(TNC, 2008)
Federated States of Micronesia (Pacific Islands)	Provide a method to sustain the long-term viability of the regionôs biodiversity. The goal was developed under the vision to have more extensive, more diverse, and higher quality of marine, terrestrial, and freshwater ecosystems that meet human needs and aspirations fairly, preserve and utilise traditional knowledge and practices, and fulfil ecosystem functions.	Two interface habitats (mangroves and estuaries), as well as a linking species (terrestrial crab) were targeted.	Not explicitly incorporated or targeted. However, one criterion to assess viability of features was the landscape context based on the dominant environmental regimes and processes that establish and maintain conservation featuresôoccurrence and connectivity. Particular to land-sea processes was the consideration of connectivity, which included access to habitats and resources needed for life cycle completion, fragmentation of natural communities and ecological systems, and the ability to respond to environmental change through dispersal, migration, or recolonization.	Major threats to key habitats (i.e., mangrove forests, estuaries, coastal freshwater marsh, freshwater rivers and streams, and high island and atoll nearshore marine) included cross-system threats: coastal erosion/sea level rise, nitrification, water pollution (mainly urban), and catchment erosion/sedimentation. No models or socioeconomic layers were constructed, but major threats to each area of biological significance (ABS) were assessed and rated by experts.	Boundaries of ABSs were delineated on maps around groups of ecologically similar featuresô occurrences. These focused on areas where nesting and co-occurrence of species and ecological systems contributed to an area's ecological complexity and integrity. Integration was achieved mainly by extending boundaries to tie together terrestrial and marine ecological systems and define ABSs that encompass entire ecosystems. The Ecoregional Portfolio Assembly Tool was used to prioritise ABSs (on the basis of a number/variety of conservation objectives and ABS functionality) and to assemble a draft ecoregion portfolio.	(MPO, 2003)
Mission Aransas Estuary, Texas (United States)	Increase the understanding of linkages between land use strategies and their effect on coastal-marine ecosystems as well as develop sound policies to maintain social, economic, and ecological values of coastal communities. The project also aimed to evaluate the current condition of the ecosystem and socioeconomic indicators, assess future trends on the basis of current policies and economic forces, and develop alternative land use strategies that meet objectives for ecological and socioeconomic indicators.	Habitats/systems and species across realms included seagrass beds, coastal live oak, dunes, wetlands, reefs, grasslands, mangroves, stream network, mammals, birds (e.g., whooping crane, plover, falcon), and reptiles (e.g., Kempôs Ridley sea turtle, terrapin turtle, snakes). In addition to "common" conservation features, socioeconomic-cultural elements were added and targeted. Some conservation elements overlapped between different environments (e.g., whooping crane habitat includes both grasslands and salt marsh).	Not explicitly incorporated or targeted. Indirectly addressed by proposing alternative development policies that will promote protection (conservation) of adjacent marine and land/freshwater systems to maintain connectivity and flows to sustain coastal processes (e.g., natural flow of water and nutrients, wildlife corridors, marsh migration).	N-SPECT was used to estimate erosion and sediment runoff (loads in stream/river mouths) from non-point-source pollution under different land use scenarios and to identify key areas that contribute to these inputs. Results were incorporated into GIS spatial analysis to model river plumes and assess (with a DSS) the potential impact of sediments on seagrass. Both land-use/build-out and river plume model scenarios were input into DSS to assess objective achievement and viability of conservation features as well as to identify areas that were adequate for mitigation (i.e., avoid urbanisation in areas close to wetlands and other areas with potential high exports of sediments if cleared).	A toolkit with four interoperating tools was used iteratively to map and evaluate current and projected land-use scenarios on marine, freshwater and terrestrial systems. CommunityViz was used to model urban growth and assess socioeconomic impacts. NatureServe Vista (DSS) was used to assess the compatibility of conservation features with land use scenarios. It incorporated the effect of water quality on seagrass based on N-SPECT and the river plume model. Land use/build-out scenarios were also inputs for the DSS. The DSS was used to assess the achievement of objectives and viability of conservation features as well as to identify areas for threat mitigation (i.e., reduce conflict and loads) and to achieve conservation objectives. Results were then exported back to CViz to evaluate socioeconomic indicators.	(Crist *et al.*, 2009; Dunton, 2009)

Northern Gulf of Mexico (United States, Mexico)	Identify sites within the northern Gulf of Mexico ecoregion that if protected would fully represent the biological diversity of the nearshore waters of this region. This would include viable populations, communities, and ecosystems.	The focus of the planning exercise was on nearshore coastal areas (land-sea interface) in particular bays and estuaries, which are shaped by terrestrial and riverine inputs. Relevant land-sea conservation features included seagrass beds, oyster reefs, salt marshes, tidal freshwater marshes, tidal flats, and submerged freshwater grasses.	Some priority sites were identified and characterised by their natural high productivity in relation to freshwater and nutrient input from rivers. Freshwater inflows were considered critical to the continued health of these ecosystems. Also considered was the natural input of sediments, which provide substrate to marshes and can retreat or diminish with the alteration of hydrologic and sedimentation regimes.	Not explicitly incorporated to modify site selection. Threats were recognised and generally evaluated for each site (bay/estuary) and then used to prioritise sites. For many sites, land-based threats (e.g., nutrient and sediments runoff) were the most important or significant stressors and were assessed in terms of their impact on priority sites. For nutrient runoff a map of hotspots (in terms of nutrient loads) of priority terrestrial areas to be managed was built, but it was not explicitly linked to site selection.	SITES was used to produce a draft portfolio that was later modified by scientists and managers. Priority sites were added to reflect variability in communities within habitats across the subregion. Design criteria included the preference to select entire bays and estuaries as landscape-scale sites (with some land components). The plan was intended to be integrated to terrestrial ecoregional assessments. This would be based on comparing high priority sites in each plan; where high-priority terrestrial, aquatic, and estuarine sites are adjacent, an overall site can be created to focus on the stresses and conservation strategies that impact biodiversity across boundaries.	(Beck *et al.*, 2000; Beck and Odaya, 2001)
Pacific Northwest Coast (Canada-United States)	Identify an efficient suite of conservation sites that will contribute toward the long-term survival of all viable native plant and animal species and natural communities in the ecoregion (including freshwater, terrestrial, and marine biodiversity).	Some targeted species, habitats, and processes incorporated land-sea linkages, including salmon (requirements for connectivity between marine waters, estuaries, and stream reaches was considered), shoreline ecosystems, supratidal/intertidal/shallow subtidal vegetation habitat types, area-based estuarine features, seabirds and shorebirds, marine mammals, and some invertebrates.	Not explicitly incorporated or targeted. However, some design criteria and adjustments promoted the integration of realms to maintain or restore connectivity between them. The study recognised that the process of integrating terrestrial, freshwater, and nearshore marine assessments can be improved by incorporating the ecological processes that link terrestrial, freshwater, and marine realms in the selection algorithm. Estuaries were mapped as polygons and used as assessment units within the terrestrial analysis.	A nearshore marine suitability index included adjacent terrestrial, freshwater, and marine factors. All adjacent lands were considered to be watersheds that directly drain into the coastal zone, and impacts were buffered and combined. Complementary analysis to the ecoregional assessment used long-term variability of sea surface temperature (AVHRR 10 year values of mean annual temperature variance) as a proxy for the river plume extent of a major river. The temperature-based proxy was compared with a salinity-based plume model (CORIE), supplemented by field observations for smaller rivers. When the projected plumes of two watersheds overlapped, cells in the overlap zone received land-based threats from both watersheds. Higher costs were assigned to marine areas potentially affected by land-based threats (i.e., units that overlapped with river plumes).	Differences between realms demanded separate assessments, but an integrated analysis was also performed to identify places where biodiversity from all realms might be efficiently conserved. The integrated portfolio was built around the terrestrial and freshwater analysis. The terrestrial/freshwater analysis (SITES) included estuaries and marine shoreline units to find efficiencies of site selection between coastal watersheds and their adjoining estuaries. The integrated terrestrial-freshwater portfolio was constructed using "vertical stacking" (i.e., selection of units is influenced by their adjacency/overlap to other units to achieve multiple objectives at once). This allows for simultaneous use of different planning unit types and assessment of realm-specific threats. The representative portfolio of nearshore and shoreline sites developed with Marxan was matched to the terrestrial/freshwater analysis to form land-sea conservation areas.	(Vander Schaaf *et al.*, 2006; Tallis *et al.*, 2008)
Palau (Micronesia, Pacific Islands)	Guide the development and implementation of a nationwide network of marine and terrestrial protected areas for Palau. Information on areas that are important for biodiversity protection and management will help guide land-use planning to ensure that development is sensitive to the terrestrial and marine environments of Palau.	Some targeted species, habitats, and processes incorporated land-sea linkages, including estuaries, coastal and riverine mangroves and streams, and crocodile critical habitat.	Not explicitly incorporated or targeted. However, two features (upland watersheds and water sources) were considered as positive "costs" in the cost layer, therefore favouring their selection. Indirectly this could have a positive effect in preventing cross-system threats or conserving land-sea processes. Selection of some habitat types was also guided by some land-sea criteria, such as targeting riverine and coastal mangroves. Streams were targeted because of their importance as corridors; for example, saltwater crocodiles use them as pathways between nesting and feeding sites.	Some cross-system threats were identified as major or important factors affecting a number of conservation features, mainly through decline in habitat quality or condition. Aquaculture (current and proposed) was the only factor explicitly included that could be considered a cross-system threat; areas overlapping with the potential area of influence of aquaculture were assigned higher conservation "costs."	Marxan was used to design five protected area network scenarios. Draft portfolios (scenarios) were reviewed to identify data gaps and solutions to fill these. Some design principles that guided the ecoregional assessment were particularly relevant to land-sea integration: give priority to areas that contain high-quality examples from multiple realms; maximise connectivity within and among ecosystems and between protected areas; include entire biological units (e.g. watersheds); and consider sea and land use, particularly proximity to threats and other protected areas.	(Hinchley *et al.*, 2007)
Prince Edward Islands (South Africa)	Design an MPA to conserve representative areas of biodiversity patterns (species and ecosystems) and processes (foraging grounds, nutrient cycles), avoid conflict with fishing, and have sensible marine management boundaries. Specific objectives included the protection of the relationship between the marine and terrestrial environments, especially of the nutrient cycle driven by birds and seals; protection of predator foraging areas; and increased resilience of species that breed on the islands or feed within the EEZ in the face of climate change.	Some targeted species, habitats, and processes incorporated land-sea linkages, including coastal processes, island shelf processes, inshore foraging areas, and seabird and elephant seal foraging areas.	The exercise explicitly targeted predators that bring nutrients to the islands (from shelf and pelagic environments) and runoff driven by precipitation that augments nutrients in shelf waters. Targeted coastal processes included: kelp beds that trap freshwater runoff and foraging grounds for nearshore foraging seabirds (defining a 1 km buffer); island shelf processes (including inshore foraging seabird areas up to the 500-m isobath); inshore foraging areas (including penguin foraging areas and phytoplankton blooms with a 40-km buffer); and nutrient input by offshore foragers (mapping foraging areas on the basis of the major movement axis of seabirds and seals).	No cross-system threats were considered. The islands are reserves. Fishing is the major threat; therefore, long-line toothfish areas (the major fishing activity) were mapped on the basis of CCAMLR Scientific Observers reports.	C-Plan was used to design the MPA following three steps: 1. identify four reserves (each representative of one of the four broad-scale habitats and the three major water bodies, and achieve as many conservation objectives as possible, including all fixed processes); 2. Connect the four reserves with straight lines to incorporate the three movement axes defined by the highest values of the bird and elephant seal habitat utilisation while also maximising objective achievement, which covers all flexible processes; and 3. assign planning units with >90% proposed in steps 1 and 2 as ñreservedò within C-Plan and recalculate irreplaceability to meet outstanding conservation objectives.	(Lombard *et al.*, 2007)

Queensland Intertidal Habitats (Australia)	Systematically identify sites for inclusion in a system of marine reserves that would protect representative examples of the full range of mainland intertidal habitats (land-sea interface features) in Queensland.	Besides intertidal habitat types, the across-shore component (habitat type) for the littoral zone higher on the shore and adjacent to each intertidal habitat type was included as additional conservation features in the reserve identification problem. Additionally, conservation features were recorded in the adjacent littoral zone.	Not considered	The length of each across-shore component of a planning unit that contained road, industry, or residential areas or other artificial feature, determined the relative cost and all other planning units had a relative cost of zero. Planning units adjacent to terrestrial protected areas were given a zero cost. This was based on the assumption that sites adjacent to residential areas are likely to have greater social or economic cost associated with the planning and management of a multiple-use marine park.	Marxan was used to construct a reserve system of coastline areas representative of intertidal habitats. Design criteria included a minimum of three examples of each conservation feature to protect against the loss of associated biodiversity arising from catastrophes (including adjacent land-use development). Incorporation of adjacent upper shore habitats in a shoreline was used to increase the likelihood that representative examples of intertidal communities and species will be conserved. For different scenarios, planning units adjacent to terrestrial protected areas were given a zero ñcostò to promote linkages. This resulted in the preferential inclusion of areas bordering on natural features in the adjacent across-shore components rather than modified sections of the coastline.	(Banks *et al.*, 2005)
Reunion Island (Western Indian Ocean)	Identify biodiversity processes at medium scale (1:100,000), map spatial components, assess their ecological status, and delineate an optimal network of corridors to safeguard their persistence.	The planning exercise focused on land ecosystems but included coastal features important for processes occurring in the land-sea interface. Conservation features included twenty native broad habitat units (BHUs) in four macro habitat units (coastal-lowland, sub-mountain, mountain, and sub-alpine), forty four BHUs (including transformed) that were surrogates for biodiversity distribution, and riverine systems (derived from 25-m DEM).	Five processes were identified and mapped on the basis of a literature review, expert knowledge, and basic GIS data. Two are relevant from a land-sea integrated perspective: 1. Processes occurring in the oceanic-terrestrial interface (e.g., fringing coastal habitat associated with cliffs, recent lava flows, and sandy beaches, some contiguous to coral reefs) such as feeding of marine birds nesting inland, arrival, foundation and evolution of plant species. **2.** Lowland-upland exchange occurring along riverine corridors, particularly important for top-down nutrient flows and bird movements.	Threats were modelled for the whole island, including those affecting the land-sea interface where some ecological processes of interest occur. The marine side of the oceanic-terrestrial interface process was considered extant (not transformed). Three predictive threat models were developed, including the distribution of urbanisation (a probabilistic, non-linear regression of 12 factors explaining urban sprawl 1989-2002), agriculture (potential areas for sugar cane and pastures identified by experts using multicriteria analysis), and plant invasions (a probabilistic, summed map of invasion threat probability and density for the 20 most invasive species using climatic envelope modelling).	Conservation corridors were delineated to encompass as much as possible of biodiversity patterns and the five processes. Marxan was used to develop a map of irreplaceability based on habitat conservation objectives. Buffers were used to map processes (i.e., oceanic-terrestrial interface: 400-m coastline buffer; riverine corridors: 200-m and 50-m buffers on each side of perennial and non-perennial rivers). Habitat transformation of spatial components of processes was calculated on the basis of the threat models.	(Lagabrielle *et al.*, 2009; Lagabrielle *et al.*, 2011)
Willamette Valleyï Puget Troughï Georgia Basin (Canada-United States)	Identify priority areas for conserving the biodiversity (terrestrial, freshwater, and marine) of the ecoregion as a guide for planners and decision makers. Develop a portfolio of priority conservation areas that are of exceptional biological value and are the most likely places for conservation to succeed based on their current condition, land use, and other factors.	Some terrestrial, marine, and freshwater targeted species, habitats, and processes incorporated land-sea linkages. Salmonids were excluded because of ecological complexity (requires connectivity design) and because they are already the focus of conservation action. Some features also served as links for (indirect) integration of priorities because they occupy more than one realm or interfaces, such as shorebirds and seabird colonies.	Not explicitly incorporated or targeted. The team agreed it would be useful to add ecosystem processes to the portfolio assembly model by including information on littoral cells in future editions or updates. Although anadromous species (e.g., salmon) were not specifically targeted, expert input to refine the aquatic portfolio resulted in the addition of areas important to salmon conservation at various life history stages as well as to provide corridors for migration.	Some cross-system threats were incorporated into the cost suitability index, but their impact (extent) on different realms was not modelled. These included dam density and dam storage (#/km^2), presence of point sources of pollution (#/km), and shoreline armouring (amount).	The terrestrial portfolio was the basis of the combined portfolio. Marine sites were added to capture areas where high-priority terrestrial and marine sites are ecologically connected. Freshwater priorities were incorporated when these overlapped with terrestrial or coastal priorities, seeking to capture areas where these systems are ecologically connected and when these added important components to terrestrial or marine sites. The nearshore marine portfolio, along with some key terrestrial areas, was locked in and SITES was run again to achieve terrestrial conservation objectives. In the construction of the aquatic suitability index, when aquatic featuresôoccurrences overlapped with terrestrial or nearshore marine portfolios, the unweighted average of the four factors was reduced by 50% (and therefore the suitability index was increased) to recognise the value of efficient conservation design.	(Floberg *et al.*, 2004)

Supplementary Material C3: Approaches, methods and components[1] considered by marine planning exercises in the Gulf of California

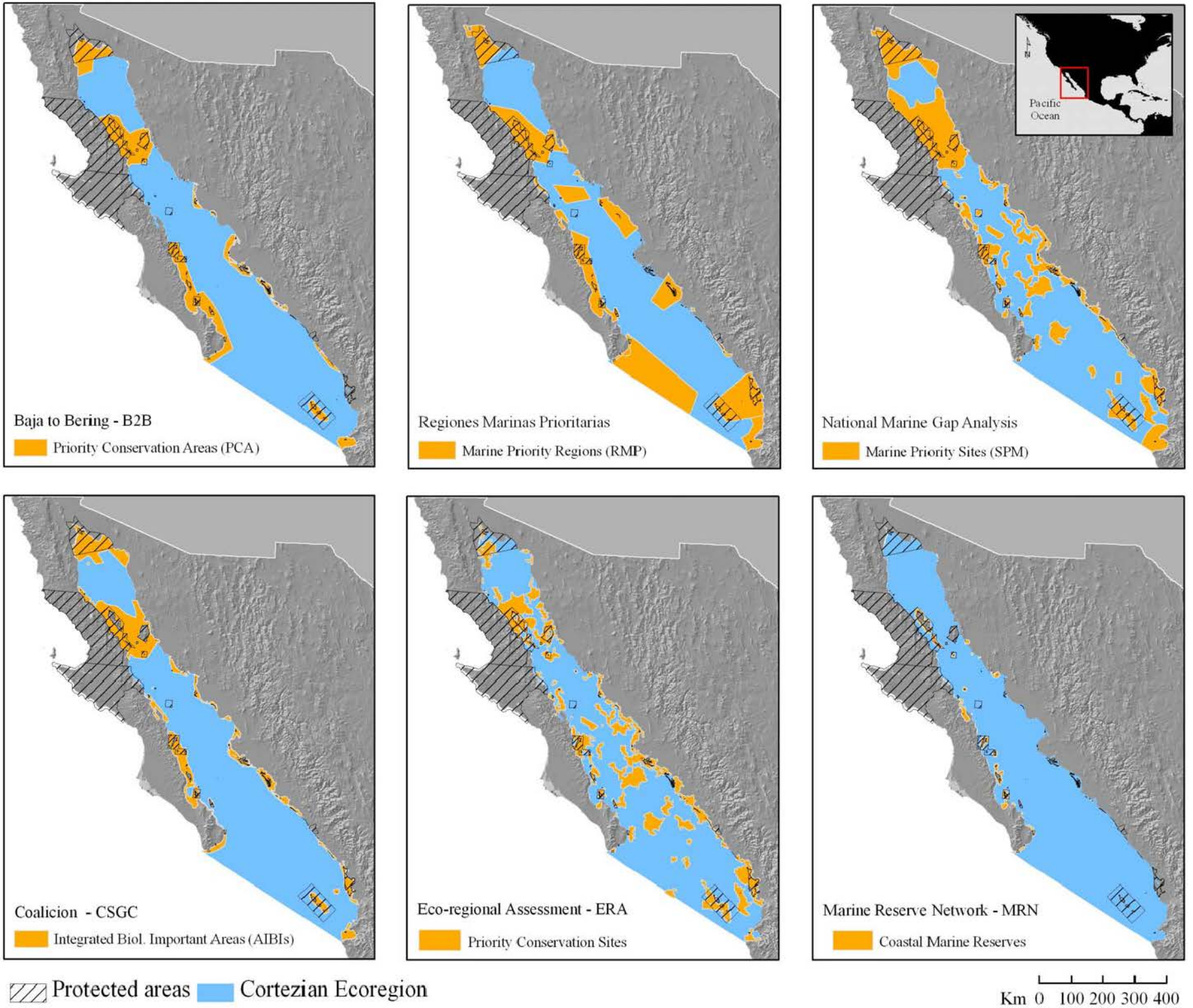
		Supra-national	National			Regional		Sub-regional
		Baja to Bering (B2B)	*Marine Priority Regions (RMP)*	*Marine Priority Sites (SPM)*	*Coalition for Sustainability (CSGC)*	*Eco-Regional Assessment (ERA)*	*BC Peninsula Priorities (BCP)*	*Marine Reserves Network (MRN)*
General approach and methods	Prioritization approach and general criteria	Ecological significance: Determined by continental endemism, high beta-diversity, significance to migratory species, and productivity. Ecologically significant sites were rated relative to others according to threats (e.g., resource extraction, pollution, coastal development) and opportunities for collaboration (e.g., existing protected status, sustainable practices, local support). Other criteria used by experts to select areas were connectivity (ecological linkages) by identifying areas important for migratory/highly mobile species and minimum area size (see ESRôs in methods).	Experts selected areas based on their knowledge and distribution maps of species and processes produced for the workshop. Stratification: Marine areas were classified in two major classes, based on physical parameters: coastal areas (from 15 meters over sea level or coastal vegetation to neritic (<200m depth) and oceanic (200m depth to limits of EEZ) zones. These types were further subdivided in 7 and 5 provinces, respectively, based on physical, geological and biological characteristics. Conservation features of interest were used as a basis for experts to select areas and agreeing on potential conservation areas by consensus; areas would ideally include most of the features.	General selection criteria included: representative and viable examples of important ecosystems/habitat types, sustainability of fisheries, high diversity, ecological processes (e.g. zones of high productivity, recruitment or reproduction), species specific habitat, connectivity, cultural, ecosystem services, and research. Selection was not based only in surrogates (information was available for several areas and issues). A coarse-scale (ecosystems/habitats) and fine-scale (endangered, endemic and key species) filters approach was followed in order to have a representation of relevant biodiversity features/components. An ecoregional approach was followed in order to represent conservation objects important in all ecoregions; a classification of sites (a posteriori) shows that representation across different geomorphology types and levels is generally good. However, priority site selection and design didnôt follow a stratified approach.	Experts selected important areas that could contain and sustain viable populations and were important or representative of selected taxa (i.e., fish, marine mammals, sea turtles, invertebrates, macroalgae, birds, land flora, land mammals, reptiles and amphibians), as well as areas where physical and ecological processes take place (viability, representation of key features and processes). These areas, defined as biologically important areas (AIB) were overlapped to define integrated important biological areas (AIBI). AIBIs were defined by an overlap of 3 or more AIBs (marine-coastal) or 2 or more (terrestrial). Design criteria also included representing important ecosystems and habitat for multiple species (wetlands) and inclusion of important ecological processes.	Ecoregional planning focus on large and functional areas, emphasizing the representation of species, communities, and systems within the ecoregion. Planning was based on a fine-filter (e.g., endangered, key, endemic, and migratory species) and coarse-filter (i.e., habitat types, key ecological/oceanographic zones) approach to selection and targeting of conservation objects. It also considers stratification of the planning domain in sub-ecoregions (stratification). The process also implies minimizing costs and proposing a spatially efficient portfolio of conservation areas. Other general design criteria considered (not implemented in DSS - Marxan) were connectivity and functionality (i.e., viability of conservation features: species, habitats and processes).	Experts selected areas based on their knowledge, previous exercises and studies. Important areas for coastal/marine conservation were prioritised based on multicriteria analyses. Preference was given to areas with: higher biological value: ecological integrity, ecosystem diversity, productivity, endemicity, richness, ecological processes, and palaeontology; more intense threats: number of endangered species, users conflict, pollution, exotic species and population growth; larger size; lower opportunities: lack of organised groups, research activity, management and spatial planning (except protected areas or previously identified as conservation priorities); higher economic importance: ecosystem services, and direct use; and presence of historical and anthropological values.	The planning exercise has a focus on reef fish, but also considered plant and invertebrate biodiversity using habitat as a surrogate. Design principles included: Protect 20% of each representative habitat and 100% of rare habitats and areas with highest species richness. Maximize protection of ecosystem functioning by protecting larval sources and nurseries for targeted fish species and by ensuring connectivity among populations through larval dispersal. Representation of major fish zoogeographic regions. Minimum reserve size to ensure self-recruitment. Minimize number of reserves to minimize conflict. Assumptions: Protection afforded by a reserve network for commercial species will ensure sufficient larval production for non-threatened species.
	Methods	The priority area selection process included four stages: 1. Draw Ecologically Significant Regions (ESR) up to 40% of EEZ and larger than 1 degree square; 2. Refine: modify or add new ESRs, keep it under 40% of EEZ; 3. Assess threats relative intensity (high, moderate or low) and trend (improving, unchanged, worsening); also rate opportunities; 4. Select up to 20% of the area within ESRs (max. 8% of EEZ), overlay proposals from the six working groups organised based on participantsô expertise (benthos, 2 for pelagic environment and 3 for planning and management) and identify degree of coincidence. Area delineation was based on baseline cartography and four general analyses (i.e., benthic complexity, oceanographic fronts, net primary productivity, and eddies). These analyses required additional GIS procedures and tools for analysis of satellite imagery.	General process: Review digital information on physicochemical factors (temperature and salinity), superficial currents and bathymetry (Data form National Oceanic Data Centre), and coastal and marine protected areas. Review and update databases from the National Biodiversity Information System (SNIB) including marine invertebrates and vertebrates. Develop thematic cartography (species distribution maps and taxonomic groupsô distribution maps per province. Discuss information and select priority regions during two expert workshops: 1. Seven expert groups: chordates (marine mammals, birds and sea turtles), fish, benthic invertebrates (corals, crustaceans, molluscs and echinoderms), oceanographic processes (physical, chemical and geologic) and pollution, fisheries, aquatic vegetation (mangroves, coastal dunes, algae, seagrasses, and wetlands) and plankton (phytoplankton and zooplankton) identified and reviewed, according to their area of expertise, priority areas in terms of biodiversity. 2. Using software to facilitate group decision-making, the areas identified in the first workshop were used to analyse use patterns and potential conflicts in the use and conservation of biodiversity, establish priorities and make recommendations.	Workshop: more than 80 persons, from 43 institutions representing academic, non-governmental and governmental organizations. Selection process: 5 groups of experts (vertebrates, benthos, plankton/fisheries, oceanographic processes, coastal vegetation) delineated priority areas, resulting maps were merged into a single national map to find level of overlap. Twenty important zones in terms of oceanographic processes were identified and used as a reference along the selection process, along with previous priority exercises and baseline cartography; finally, a gap analysis to evaluate coverage of current MPA system of priority sites identified in this exercise was developed. Conservation objects were used as a reference for experts to identify priority areas, which included most of these objects.	Data collection and preliminary analysis: Gathering, organizing, and preliminary analysis of datasets (update species inventories). Participative workshop: Seven working groups: biodiversity (land flora, land fauna, marine biota, and wetlands), processes (marine physical processes, marine ecological processes, and socioeconomic processes). Working groupsô tasks included integrating available (spatial) information, generating updated inventory of species, selecting preliminary conservation objects, and mapping the most suitable locations of conservation objects. Selected conservation features were subject to external peer-review. Integration of AIBs for taxonomic groups and ecological and physical processes (overlap) to define integrated important biological areas (AIBIs). Areas were reviewed based on their geographical limits, the relationship to other biologically important areas, and the ecological and physical context and their boundaries further refined. Develop maps of anthropogenic threats to biodiversity and potential social conflicts.	The planning exercise followed the general principles of ecoregional assessments (Groves et al. 2000): 1. Collection of biodiversity and socioeconomic spatial data. 2. Stratification in sub-ecoregions (based on sea surface temperature model, using monthly (1982-2000) AVHRR 9Km satellite imagery derived, NASA). 3. Identifying and set conservation targets (i.e., objectives) for coarse scale/filter (habitat types, ecological systems, communities) and fine scale/filter (rare, endemic and endangered species, rare communities, aggregations) conservation objectives. 4. Identify and evaluate conservation "cost" in terms of relative level of threat (summed individual threat layers) for each planning unit. 3. Construction of a portfolio supported by a DSS (Marxan) based on defined targets, while minimizing ñcostsò. 4. Refine the portfolio based on expert input.	The planning process comprised 3 stages: 1) compilation of information about general problems in the use and conservation of marine resources; identify important conservation areas, and define assessment criteria for areas and actions based on literature and expert opinion (interviews); 2) multidisciplinary workshops to modify and agree on a list of problems (and causes) and to propose conservation/management actions; and 3) expert-based prioritisation of actions and areas based on qualitative multicriteria analysis. Areas were Priority areas are associated with actions directed to improve marine resource management and conservation, including: regionalised fisheries management; ecotourism and use of islands; coastal zone use; and protected areas.	Automated selection process based on DSS. SITES/SPEXAN: parameters: 1,000 runs, each 100,000 iterations, maximization algorithm to select single best solution (meets all conservation targets and minimize habitat area and social conflicts, and maximize species richness and connectivity). Total boundary length (perimeter) of the system was not minimized (BLM = 0). Species richness and fishing pressure were weighted (0.3 and 0.7, respectively). Connectivity: Select portfolios with distance between adjacent reserves not exceeding 100km (Euclidean minimum distance between nodes). Minimum size: Large planning units and large area. Stratification: The study area was subdivided in three zoogeographic regions following fish abundance data (CCA) and the selection model was replicated in each of these. Rare habitats had a penalization factor greater than other habitats.
	Land-sea integration	Experts considered cross-system threats, such as land-based pollution, as well as coastal land-change in the process of selecting priority conservation areas, but not explicit criteria to deal with these threats are reported (e.g., avoidance of these threats or prioritizing areas affected by these threats). No explicit land-sea connectivity is considered, either with freshwater or terrestrial areas. Some areas with terrestrial and interface components, such as lagoons, estuaries and islands, were selected.	No explicit land-sea connectivity was considered in the prioritization/design process. However, some selected areas have terrestrial and interface components, such as lagoons, estuaries or mouth rivers. In the threats analysis some cross-system were identified, such as land-based pollution.	No explicit land-sea connectivity was considered in the prioritization/design process. However, some selected areas have terrestrial and interface components, such as lagoons, estuaries or mouth rivers. Some considerations (not spatially explicit) to cross-system threats, such as land-based pollution and land-sea processes, such as nutrient river flows that sustain coastal and marine systems were given in the selection process. It is expected to integrate terrestrial, freshwater and marine priority areas in a second phase to the gap analysis.	Simultaneous identification of priorities for terrestrial and coastal-marine areas. Some wetlands also include river components. Some land-based threats where considered, but these were not explicitly incorporated in the priority areas selection process. It was recommended that areas where biological importance and anthropogenic pressure coincide must be conserved not only to maintain biodiversity but also to assure a sustainable economy.	No explicit land-sea connectivity criteria or models were incorporated in the design, either in relation to freshwater or terrestrial areas. However, some selected priority areas included terrestrial and interface components, such as lagoons, estuaries, and others also their associated islands. Terrestrial limits included coastal wetlands (0 m altitude). In terms of cross-system threats, aquaculture (shrimp) farms, population development and ports (oil) were incorporated through single or multiple buffers associated to different weights (costs) in the general cost layer.	No explicit land-sea connectivity was considered in the area selection and prioritization process. However, some priority areas have terrestrial and interface components, such as lagoons and estuaries. Cross-system threats, such as land-based pollution and damming, were considered by experts in the prioritisation process.	No explicit land-sea connectivity criteria or models were incorporated in the design. However, some targeted and selected interface habitats, such as mangroves, were considered as critical for targeted species (fish), therefore dependant on land-sea connections.
Regional scoping	Define planning domain	Administrative and ecological boundaries: North American Marine Ecoregions (Wilkinson et al. 2009) and EEZ of Canada, United States and Mexico (only Pacific Coast). Northern and southern limits of North Americaôs marine ecoregions were defined by experts according to oceanographic features and geographically distinct assemblages of species. Planning included estuaries and islands, but no upland areas or freshwaters. Stratified by marine ecoregions (defined based on temperature, currents, etc.).	Administrative, physical and ecological boundaries: Mexico's EEZ; terrestrial limit was defined based on altitude (15 m above sea level), where halophytes and coastal wetlands are located, as well as by the presence of mangroves and dune vegetation. Vegetation distribution maps followed the national vegetation maps (INEGI 1:250,000). The general scale of the study is defined by national baseline cartography (1:4,000,000).	Administrative, physical and ecological boundaries: Mexico's EEZ. The planning exercise included estuaries, islands and coastal vegetation, but inland limit was not clearly defined. The scale of baseline and thematic cartography used to delineate the priority regions varied from 1:250,000 to 1:10,000,000.	Ecological and administrative boundaries: The planning domain is limited to the Gulf of California and west coast of the Baja California Peninsula (200km buffer). The domain also incorporated terrestrial components, limited by the boundaries of the five states bordering the Gulf of California (Nayarit, Sinaloa, Sonora, Baja California and Baja California Sur).	Ecological and physical boundaries: Gulf of California Ecoregion (Beck et al. 2003) and southwest coast of Baja California Sur, up to the Californian Ecoregion (Gleason et al. 2004). Limited by the Upper GoC and Colorado River Delta (northern limit) and Cabo Corrientes, Jalisco (southeast limit) and the southern portion of Baja California Peninsula (Cabo San Lucas) to Punta Abreojos, Baja California Sur (western limit). Inland boundaries were defined by the mean sea level (0 metres above sea level) and coastal wetlands. Coastline was defined based on 186 merged digitized maps (INEGI 1:50,000). Stratified in Marine Ecological Units (defined based on temperature, depth, continental flows, and species distribution).	Ecological and management boundaries: The planning was restricted to coastal/marine (Gulf of California and Western coast of Baja California Peninsula) and terrestrial components (BC Peninsula). The coastal zone was defined as the land portion affected by its proximity to the sea and vice versa, including islands, coastal lagoons, estuaries, wetlands, bays and marine zones under the jurisdiction of the states of Baja California and Baja California Sur. Previously selected priority regions (RMP) served as a geographic framework.	Ecological boundaries: The study area covers a large portion of the Cortezian (Gulf of California) marine ecoregion, but is limited by its focus on the zoogeographic areas relevant for rocky coastal habitat and associated reef fish (~1000km of latitude) and restricted to planning units which only cover coastal areas (see planning units). The domain was subdivided in three sub-regions defined by the limits between zoogeographic regions based on the ordination of sites in the canonical correspondence analysis on fish abundance data for boulder habitat in the Gulf of California.

[1] Planning components are grouped in 3 stages (regional scoping, integration and analysis, and synthesis) based on the framework by Alvarez-Romero et al. (2011): Integrated land-sea conservation planning: the missing links. *Annual Review of Ecology, Evolution, and Systematics* 42: 381-409.

	Identify and involve stakeholders	The planning exercises was leaded by the CEC and supported by the Marine Conservation Biology Institute (MCBI). It was based on a workshop with the participation of governmental agencies (e.g., US Department of Commerce, NOAA, CONANP), NGOs (e.g., TNC, NRDC, CI WWF, Pronatura, Living Oceans Society, Parks and Wilderness Society, Ocean Wilderness Network, Surfrider Foundation, Ecotrust), Universities (e.g., Simon Fraser, McGill), and some natural resource users representatives (i.e., fishermen). Major inputs from participants included selection of conservation objectives, data collection, selection and delineation of priority areas, and documentation of priority areas and islands.	Coordinated by Mexicoôs National Commission for the Knowledge and Use of Biodiversity (CONABIO), the planning process followed an expert-driven approach. Participants in the workshop included public and private research institutions (UAM, UABC, UABCS, UDC, UNAM, COLMEX, IPN, ITESM, CICESE, CIBNOR, CICIMAR, INECOL, ECOSUR, CRIPs-INP), governmental agencies from environmental, fisheries and tourism sectors (e.g., DGVS-SEMARNAT, CONANP, PROFEPA, INP, SAGARPA, CONAGUA, SECTUR), NGOs (WWF-Mexico, Wetlands international, Fundacion ARA), MPA managers (e.g., Isla Contoy), international organisations (i.e., CEC). Main inputs from experts were identifying and delineating areas were conservation features of interest occur, potential conflicts in the use and conservation of biodiversity, defining the priority status, and formulating general recommendations.	Coordinated by Mexico's National Commission for the Knowledge and Use of Biodiversity (CONABIO) and supported by the National Protected Areas Commission (CONANP) and two NGOs (TNC, Pronatura). Expert-based (participatory workshop) approach. Around 80 people with expertise in different fields (e.g., coastal and marine biodiversity, conservation, ecology, oceanographic processes, natural resource management) and regions of Mexican coastal and marine ecosystems participated in the workshop. These included researchers (e.g., UNAM, CINVESTAV Merida, ECOSUR Chetumal, UABC, UABCS, UDG, UV), national and regional environmental NGOs (e.g., CI, GECI, Pronatura Noroeste, Terra Peninsular) and governmental agencies (e.g., SEMARNAT-DGPAIRS, DGVS, INE, INEGI). Coordinators defined the general goals, planning domain and general methodologies, which were validated by the experts. Major inputs from participants included additional data, selection of conservation objectives, selection and delineation of priority areas, threat assessment, and documentation of priority areas and islands.	Highly participative process with international, national, regional and local stakeholders. The planning process was coordinated by the Coalition for Sustainability of the Gulf of California (CSGC). The CSGC included regional conservation organizations, federal and state governments, natural protected areas, and public and private universities and research centres. The selection of priority areas was based on a participatory and consensus-based process, where diverse stakeholders provided information and participated in the analysis and portfolio design. The process included NGOs (e.g., Pronatura Noroeste, Niparaja, COBI, WWF-Gulf of California, CI-Mexico), research centres (e.g., UNAM, UABC, UABCS, CIBNOR, CICESE, Tec de Monterreyôs CECARENA), international organizations (e.g., CEC), Marine Protected Areas managers, federal and governmental agencies (e.g., CONABIO, INE, SEMARNAT, CONANP, State Agencies), industry (e.g., CEMEX), and other organisations (e.g., FMCN). Participants in the workshop identified major goals, set conservation objectives, compiled and analysed information, developed threat/conflict assessment and selected and delineated the priority conservation areas.	The process was developed by Comunidad y Biodiversidad (COBI) and The Nature Conservancy (TNC) with the support and input from experts, managers and diverse organizations. The planning team defined the general goals, objectives, and methodology, and based on these developed the priority areas portfolio. The major input from natural resource users (i.e., fishermen) was information on fishing areas via interviews. During a workshop the planning process and portfolio was presented for an external group of experts to review the plan, provide additional information (gaps) and propose potential adjustments to the portfolio. Participants to the workshop included international, national and regional NGOs (CI-Mexico, WWF-Mexico, Niparaja, Pronatura Noroeste), research institutions (UNAM, UABC, UABCS, UV, UDG, CICIMAR, SCRIPPS, Moss Lab, Blue Ocean Institute), protected area managers (e.g., Loreto Marine Park) and national and US governmental agencies (e.g., INE-SEMARNAT, NOAA, NMFS, NOAA ï Southwest Fisheries Science Centre).	Diverse stakeholders participated in different stages of planning. First, natural resource users (fishermen, tourism), industry, tradesman, businessmen, governmental agencies (different areas and levels), MPA managers, researchers and conservation organisations were interviewed regarding resource uses (types, methods, areas and regulations) and users (organisations, conflicts). Stakeholders also attended workshops to review problems, and to propose and identify priority actions to address them. A selected group of researchers identified and prioritised important areas for marine/coastal conservation.	The planning exercise was basically and academic process organized by two research centres (Scripps Institution of Oceanography and UABCS) and one NGO (WWF-Mexico). Input from artisanal fishermen was important for mapping fishing areas.
	Evaluate conservation context	No assessment of conservation context was undertaken before the prioritization exercise.	No assessment of conservation context was undertaken before the prioritization exercise. The conservation context was based on the experience and knowledge of the experts.	A general assessment of the factors that influence marine conservation in Mexico was undertaken and used as a reference for the planning process. These included conservation and management actions/programs in place (e.g., MPA system, international nominations, previous planning exercises and initiatives; a major objective of the exercise was to develop a gap analysis), ecoregional framework (framework for ecological analysis and regional, national and supra-national cooperation), biological/ecological (productivity zones, biodiversity), physical (e.g., oceanography, geomorphology) and insular biodiversity (update and digitize cartography and data, existing conservation organisations and programs). A broad evaluation of the political and socioeconomic context was undertaken.	A general assessment of the regionôs factors that influence conservation in the region was undertaken previous to the workshop. This was mostly focused on compilation of information through a participative process with experts about biological/ecological (productivity zones, biodiversity) and socioeconomic (e.g., population, tourism, agriculture, fishing and aquaculture, coastal development), as well as the protection/management in place (e.g., MPA system, conservation initiatives).	A general assessment of the regionôs factors that influence conservation in the region was undertaken and used as a reference for the planning process. This included: Physical (e.g., climate, oceanographic phenomena, geomorphology), biological/ecological (productivity zones, biodiversity) and socioeconomic (e.g., population, tourism, agriculture, fishing and aquaculture, coast development and modification, exotic species), as well as the protection/management in place (e.g., MPA system, international nominations, previous planning exercises and initiatives).	A detailed assessment of focal areas was the first stage of planning and included: account and status of biodiversity; use of coastal/marine resources and associated problems; and existing regulations, policies and plans for environmental protection, natural resource management/use and development. The assessment also included a regional socioeconomic evaluation of fishing, tourism, agriculture, industry and coastal development. Assessments were based on a broad literature review and interviews to stakeholders. The assessment included identification of threats and opportunities for conservation and sustainable use.	No assessment of conservation context was undertaken before the prioritization exercise.
Integration and analysis	Define goals	Biodiversity conservation, resource management, cultural values, recreation, and research priorities: Conserve biodiversity as well as provide benefits to fisheries, cultural values, recreation and scientific research. The process aimed to define where conservation action is immediately necessary (course for future conservation alliances and action), as well as the basis to implement an integrated network of marine protected areas in North America. Specific goals included: identify regions of high ecological significance at continental scale (ESR); within ESRs identify priority areas for conservation action (PCA) considering threats and opportunities for international collaboration; and document each PCA.	Biodiversity conservation, resource management, and research priorities: Identify areas important for their biological diversity, the use of natural resources, or the lack of knowledge about biodiversity, as well as identify the major threats to these and suggest general prevention, mitigation or control strategies. Also identify areas where further research/data is needed.	Biodiversity conservation, resource management, and restoration: Identify areas where conservation efforts should be focused; evaluate and improve Mexico's coastal-marine and deep ocean protected area system (Gap analysis) to define in situ conservation priorities in order to guide the creation of new protected areas and promote their conservation through other instruments, including integrated coastal zone management, sustainable use, fisheries management, and restoration.	Biodiversity conservation: Identify, through a highly participative process, the critical areas for biodiversity conservation and determine the relative level of anthropogenic pressure on natural resources, as well as the potential for social conflict derived from conservation actions throughout the Gulf of California region.	Biodiversity conservation and natural resource management: Promote a regional focus in marine-coastal conservation and management; provide a detailed portfolio of priority areas that represent the diversity and distribution of species, natural communities, and ecological systems of the ecoregion. Also, contribute to the knowledge of biodiversity of marine and coastal environments, and facilitate the definition and implementation of conservation strategies.	Biodiversity conservation, resource management, livelihoods, and research priorities: Identify problems (and causes) in the use and conservation of marine resources as the basis to identify and prioritise actions to solve these problems, promoting a better use of marine and coastal resources, and as a consequence improve livelihoods of communities within the region. Simultaneously, identify and prioritise areas for the conservation of coastal and marine resources of Baja California Peninsula. In addition, planning also aimed to develop a robust and flexible method that can be updated and reproduced at different scales.	Academic/Biodiversity conservation and resource management: Develop a method (model) to design marine reserve networks to protect biodiversity and complement fisheries management by using multiple levels of information on biodiversity, ecological processes (spawning, recruitment, and larval connectivity), and socioeconomic factors to reduce conflict, and taking into account connectivity.
	Set conservation objectives	Qualitative conservation objectives regarding both patterns and processes were set. These included maximizing the selection of complex benthic habitats, connectivity, beta-diversity, and protection of umbrella and charismatic species, as well as including important endemic species, key and vulnerable habitats, critical habitats for species of common concern, and important high productivity areas.	No conservation objectives were set The RMP are a framework to be considered by different sectors (e.g., agencies, scientists, NGOs) for the development of biodiversity conservation, use and research plans.	Qualitative conservation objectives were set to include areas with: high species richness and habitat of endangered, endemic or protected species; important oceanographic processes; high abundance of plankton and important for fisheries (socioeconomic value); and relevant coastal habitat/vegetation systems. These included features and processes occurring in the coastal zone, continental margin and oceanic zone.	Qualitative conservation objectives were set to include important areas that could contain and sustain viable populations/systems or were important and representative of selected taxa (i.e., fish, marine mammals, sea turtles, invertebrates, macroalgae, birds, land flora, land mammals, reptiles and amphibians), key species (mangroves and seagrasses), and key ecosystems (wetlands). Conservation objectives also included incorporating important oceanographic and ecological processes.	Quantitative conservation objectives were set by the planning team, taken into consideration current distribution, data confidence/uncertainty (model vs. current occurrence) and conservation viability. Targets varied between different conservation features: Species: 30%-40% of important areas for sea turtles, sea mammals, seabirds, and fishing bat; 100% of distribution range for endangered endemics; 10% for most species occurrences and 20% for vulnerable species. Representation units/habitats: 20% of sand beaches, rocky shores/cliffs and dunes, 80% of fine sediment beaches, 30% of benthic complexity, and 100% for mangroves, wetlands, swamps, and coral reefs. Special elements: 50% of Sargasso meadows, 30% of sardine recruitment areas, and 100% of seamounts, rhodolith beds, seagrass meadows, sea turtles nesting areas, seabird nesting colonies and sea lions reproduction colonies). Processes: 30% of primary productivity and upwelling zones. Other: 30% of MPAs and selected ethnic exclusive zones.	General qualitative conservation objective: identify natural areas of particular relevance for biodiversity conservation in the region. Areas were then prioritised based biological value (ecological integrity, ecosystem diversity, productivity, endemicity, richness, and outstanding ecological processes, such as migration and species aggregations) and threats (areas with more endangered species).	Qualitative and quantitative conservation objectives were set based on ecological and marine reserve design theory. Targets varied between different conservation features: Species: Maximize reef fish diversity (based on mean species richness/planning unit, obtained by GIS Kriging interpolation). Representation units/habitats: Theory suggests that marine reserves should protect >20% of the habitat to enhance fisheries. Special elements: 50% of mangrove fish nurseries, XX % of larval sources (sites), and 100% of fish spawning aggregations (irreplaceable sites), coral reefs (rare communities), and seagrass beds (critical habitat). Processes: Maximize protection of ecosystem functioning by protecting larval sources and nurseries for targeted fish species and by ensuring connectivity among populations through larval dispersal.

Biodiversity patterns	Species: High extinction risk, protected/listed, feasible to recover/manage, flagship, and/or endemic to the B2B region or areas within (e.g., Totoaba macdonaldii and Phocoena sinus in the Upper GoC). Focal species: Transboundary, migratory, umbrella and keystone species (to target critical breeding or feeding habitats for multiple species). Representation units/habitats: Benthic complexity (indicator of heterogeneous habitats, associated with species richness). Special elements: Deep sea corals (habitat-forming species and indicators of areas that can provide critical fish habitat); seamounts (high diversity, endemism and productivity, rare and delicate ecosystems). Other: Beta-diversity (experts considered changes in habitat and biological diversity across the ecoregion and within each sub-ecoregion). Spatial data used to represent targeted features included high resolution bathymetry to identify slope changes (benthic complexity model), critical habitats and broad distribution maps (species), and occurrences (deep sea corals, seamounts).	Species: Polychaetes, molluscs, sea turtles, crustaceans, echinoderms, macroalgae, fish, marine mammals, resident and migratory birds, and corals (no selection criteria, based on available information). Representation units/habitats: Aquatic vegetation: mangroves, coastal dunes, algae, seagrasses, and wetlands (high-diversity ecosystems, critical habitat for many species). Special elements: Phytoplankton and zooplankton (areas of high productivity). Other: Important fishing areas. Spatial data: Based on species records (SNIB database) different thematic maps were produced, including: species distribution maps (1:1,750,000), taxonomic groups distribution maps by coastal and oceanic provinces (1:1,000,000 and 1:1,750,000). Aquatic vegetation ecosystems were mapped based on 1:250,000 cartography. No information is available on the spatial data for plankton and fisheries.	Species: Vertebrates and invertebrates (focus on endangered, endemic, micro-endemic, and protected species, but all species records were analysed). Representation units/habitats: Coast types, coral and deep reefs, seagrass meadows, kelp forests, wetlands, and benthic complexity (areas of high species diversity and ecological relevance). Special elements: Feeding, reproduction and development areas, nesting beaches, multiple species refugees, fish concentrations, islands, seamounts, rift zones, bacteria mats, minimum oxygen zones, top predators concentration areas. Spatial data: Important areas for conservation objects were delineated (mapped) by experts based on personal knowledge and supported by species data (153,242 occurrences for 9,253 species, 1,036 infra-species; including 197 protected and 75 endemic spp.) provided by experts during the workshop, data contained in SNIB (Mexico's biodiversity database) and literature. Other maps used included: vegetation types and land-use (INEGI 1:250,000). Existing models from previous planning exercises and research projects were also used, including benthic complexity, habitat and coastline types, and special features (e.g., seamounts).	Species: Selected species grouped and analysed by taxonomic group.- 27 fish, 8 mammals, 6 sea turtles, 213 invertebrates, 63 macroalgae, 56 birds, 214 land flora spp., 36 land mammals and 37 reptiles and amphibians (considered key biodiversity elements, selected based on population and habitat condition, endemism, distribution ranges, threats and trends, from high threat to stable). Focal species: Marine grasses and mangrove species (overall importance as ecosystems). Special elements: Wetlands (important ecosystems and habitat for multiple species). Spatial data: Maps for selected taxa (species) and key ecosystems were delineated by experts based on individual knowledge and supported by point data (occurrences) from collected information (a geo-database was built: Prometeo). These maps represented important areas where suitable populations of selected taxa and key ecosystems may exist.	Species: Important areas and occurrences of sargasso, seagrass, fish, corals, sea turtles, sea mammals, and seabirds (i.e., protected, threatened, endemic, key, migratory, umbrella, specialists, charismatic). Representation units/habitats: Mangroves, coastal wetlands and swamps, fine sediment beaches, sand beaches, rocky shores, cliffs, and dunes, benthic complexity, and coral reefs. Special elements: Seamounts, rhodolith beds, sargasso and seagrass meadows, sea turtles nesting areas, sea lions and seabird reproduction/nesting colonies, sardine recruitment areas. Spatial data: important species areas polygons (aggregated occurrences/planning unit interpolated to 1km; only units with a min. #records); important invertebrate areas (from CSGC); spp. occurrences (~45,900 records); costal types: linear features and mangroves derived from digitized 1:50,000 cartography and verified with Landsat imagery; benthic complexity model (see Ardron 2002); coral reefs (point records); sardine polygons (fishermen interviews); ethnic groups (studies), MPA polygons (CONANP); and species records (agencies, NGOs and research centres databases).	Spatial data on biodiversity were not used to identify the priority conservation areas. However, the following biological criteria were used to prioritise the selected areas: ecological integrity, ecosystem diversity, endemicity, richness, and number of endangered species.	Species: Species richness of reef fishes on shallow rocky bottoms (boulders and walls, 5 to 20m). Representation units/habitats: Sandy bottoms, seamounts, rhodolith beds, sargasso beds, black coral beds (representative habitats, used as surrogate for plant and invertebrate biodiversity) and rocky habitat (boulders and walls are important reef fish habitat). Special elements: Mangrove fish nurseries (habitat for critical life stages in fish life cycle, important for commercial fish), larval sources (sustain species and fisheries), spawning aggregations (targeted by fishermen, important areas for 7 commercial fish species), and coral communities and seagrass beds (rare habitats). Spatial data: fish species richness maps were constructed for all habitats, estimated based on scuba sampling (134 identified species) and Kriging interpolation. Fish nurseries (mangroves) were identified by scuba sampling for juveniles and Landsat MSS 1992 images. Important larval sources (interviews to local fishermen); spawning aggregations (surveys and interviews with fishermen; focused on big commercial fish species that spawn at specific locations and that are targeted by fishers).
Biodiversity processes	Oceanographic and coastal processes associated with areas of important biological diversity, ecological significance for targeted species or high biomass and productivity. These included oceanographic fronts (critical habitat for fish, sea turtles, seabirds, and marine mammals; proxy for high secondary productivity), currents, gyres and eddies (related to upwellings, trophic cascades and plankton blooms), net primary productivity (biological richness, fisheries), and migration corridors (critical areas for feeding, calving and breeding). Processes were mapped by means of models: sea surface temperature frontal density (fronts), altimetry or sea surface height based on satellite imagery (cold-core eddies, only Alaska), estimated chlorophyll, photo-synthetically available radiation and SST (productivity), and tracking of whales and sea turtles, associated with fronts (corridors).	Oceanographic processes considered by experts in the delineation of priority areas included physical, chemical and geologic (biological and ecological importance, high productivity). Maps of these processes were produced for the workshop. No information is available on the spatial data used to map or model processes.	Oceanographic and coastal processes considered by experts in the delineation of priority areas included: upwelling zones, oceanic fronts, vertical mixing, advection, gyres and eddies, currents, and river discharges. Processes were selected based on their relation to high productivity areas, biological richness, connectivity among habitats such as estuaries, mangroves and coral reefs, regional balance in the ocean-atmosphere gas exchange, or function as corridors for nutrients, larvae and organisms between coastal ecosystems. Zones were identified by experts based on their research (maps/models); no surrogates or proxies (e.g. sea surface temperature, chlorophyll) were directly used to map these processes for the workshop. However, existing models from previous planning exercises and studies were also used.	Experts delineated areas where oceanographic processes relevant for biodiversity occur, including upwelling, oceanic fronts, and currents. They also delineated areas with important ecological processes, where main interactions and connections among species from different taxonomic groups, and also between these and physical processes occur. Working groups also considered aspects such as reproduction, larvae dispersion, birth rates and recruitment, and migration in the identification of marine ecological processes.	Primary productivity (biological importance, productivity indices integrates cumulative effects of many processes and interactions) and upwelling zones (biological importance, diversity, input of energy into the system). Processes were mapped based on satellite imagery analysis: Productivity zones were derived from daily MODIS (1 year, 2003-2004). Upwelling zones were defined based on AVHR derived seasonal isotherms (Winter: 16ÁC and 18ÁC; Summer: 29ÁC) that determine the occurrence of upwellings in the region.	The presence of ecological processes were not explicitly used to identify priority areas, but were incorporated into the biological criteria used to prioritise areas. Processes considered by experts included: primary productivity, upwellings, migrations and biological corridors.	Population connectivity (fish and invertebrates): Not explicitly mapped or modelled. Incorporated though general design criteria and implemented in Avenue (Avenue: VKMax.AVX) via reserve selection rule (constraint) - Distance between adjacent reserves (Euclidean minimum distance between nodes) should not exceed 100km to maximize connectivity.
Threats	Assessment based on expert opinion. Experts were asked to rate ESRs according to their knowledge of regional threats, including: organic and inorganic pollution form land-based and sea-based activities (i.e., offshore oil, military bases, nuclear plants, forestry/deforestation, shipping, urban run-off, industrial activity, nutrient loading); fishing (i.e., subsistence and commercial fishing of fish and invertebrates, recreational fishing, and trawling); hunting (marine mammals), aquaculture (includes risk from introduced species); damming (i.e., reduced freshwater flow, power plants); tourism, coastal development and urbanization (for the GoC based on Escalera Nautica project, which included development of harbours, wharves, hotels, restaurants, airports and airstrips); physical alteration of coastlines (i.e., development of estuaries and deltas, modification of marshes, dredging, ports, nautical projects); and damaging recreational use (i.e., tourism, marine recreation). Where applicable, experts ranked threats relative intensity (high, moderate, low) and assessed the current trend (improving, unchanged, worsening).	Threats were not explicitly and systematically used in the priority regions selection process. Once the priority regions were defined, the major threats to biodiversity were identified and documented for most areas. These included: overexploitation, pollution, petroleum industry, tourism, and lack of strategies/integrated management. No specific spatial data (e.g., cartography, models) is reported; threats were assessed based on the knowledge of experts and literature.	Not explicitly incorporated. Threats were analysed after delineating priority sites, and therefore not used as a criteria for the selection of these.	Threat was one of the general criteria considered by experts to delineate viable priority conservation areas. Maps of anthropogenic threats were estimated for current and near future (5 year scenario) and constructed in a workshop. A spatial index of anthropogenic pressure on biodiversity was calculated based on the sum of all threats (population, tourism, agriculture, aquaculture, fisheries). Each threat was assigned with a value: 1 (low) to 5 (high) according to estimated impact. Total scores (summed map) were standardized to fit in a 1ï 10 scale; trend maps were constructed based on score difference between present and near-future scenario (5 years); for land-based marine pollution point sources were buffered. The final anthropogenic pressure map covers the terrestrial and marine realms. The threats map was not integrated or used as a guide to adjust or modify the selection of priority conservation areas.	A cost index was constructed based on summed layers (values) of individual threats including: Tourism: Data from Escalera Nautica Project (ENP). Two types defined based on potential impact: selective (ENP) and massive (all other touristic sites). Costs were assigned to sites through buffers: selective (1 ring - 3 Km, 893) and massive (3 rings - 10, 30, 50 km; 2678, 1785, 893). Population: Based on marginality (INEGI) and population size, different number of rings (impact buffers) and "costs" were associated to populations (i.e., small (<10,000 people) - 1 ring (1785); medium (10,000-100,000) - 3 rings (2678, 1785, 892); and large (>100,000) - 4 rings (3571, 2678, 1785, 892). Aquaculture: Potential impact derived from shrimp farms, cost: 143. Fishing: Costs were assigned according to potential impact of different fishing types: recreational (143), molluscs (4167), shrimp: artisanal (208) and commercial (333), sardine (167), others (375), shark (333), crustaceans - crayfish and lobster (4167), squid (NA). Navigation routes: Potential impact caused by pollution originating from ships; a 1km buffer was defined for oil (cost: 893) and commercial transport (cost: 2678) routes. Ports: Fishing and oil ports were included independently: 2 rings of different potential impact assigned to both types of ports (2678 and 892).	Based on a literature review and interviews, regional threats and problems/conflicts regarding conservation and use of marine/coastal resources were identified. These were related to fisheries, coastal zone use, ecotourism and use of islands, and natural protected areas. Threat assessment was undertaken independently for each coast of the BC Peninsula and is presented as causes-effects, emphasizing the need to address root problems, not necessarily through spatial actions. Problems/threats categories (and examples of symptoms) included: access to areas/resources (illegal fishing, overexploitation, conflict, land clearing); management bodies (inadequate planning, regulation and enforcement, corruption, jurisdictional fragmentation and overlap, poor socioeconomic, fisheries and biological data); market (resources/services valuation, poor incentives and marketing); economic recession (migration); fisheries organisation (illegal enterprises, poor infrastructure); research-fishing management gap; illegal foreign fishers.	Not considered. See costs and opportunities.
Ecosystem services	Not explicitly targeted. The value of some PCAs to provide key ecosystem services was recognized. For example, coral reefs and kelp forests that help protect adjacent sandy beaches or rocky coasts from wave action, estuaries that provide nutrients to more offshore sites, and marshes that help to purify water from land-based sources.	Not explicitly targeted. However, conservation features of interest considered by experts in the delineation of priority areas included areas identified as important for phytoplankton, zooplankton and fisheries.	Not explicitly targeted. Included as conservation objective (i.e., high abundance of plankton and important for fisheries), but not explicitly mapped, modelled or targeted.	Not explicitly targeted. Some ecosystems were recognised for their importance in delivering services (e.g., mangroves as nurseries of commercial fish species). One major conclusion of the exercise was that areas with high biodiversity have high socioeconomic importance due to the flow of ecological services and goods provided to the economic system.	Not explicitly incorporated. However, some conservation features were also targeted because of the services these can provide (e.g., mangroves, coastal wetlands and swamps provide coastal protection, seagrass meadows are potential sources of human and animal food)	Not explicitly targeted. However, experts considered the ecosystem services in the prioritisation of conservation areas (e.g., area importance as habitat or reproduction zone for commercial species, energy-exporting centre, and protection of the coastal zone against erosion or natural disasters.	Not explicitly targeted. However, some targeted ecosystems and features were recognized for their importance for commercial fisheries (e.g., mangroves as fish nurseries, fish spawning aggregations and larval sources).
Climate change	Not considered	Not considered	Not considered	Not considered	Not considered	Not considered	Not considered
Explicit costs	No costs were incorporated	No costs were incorporated	No costs were incorporated	Social, economic, cultural or political costs are mentioned as part of the criteria utilized in order to define current and near-future social conflict potential. A spatial index of social conflict potentiality regarding the use of natural resources in the region was developed. The index represented different levels of social conflict potential, which is an indication of the feasibility (or difficulty) of implementing conservation programs or protected areas, but was not used to delineate priority sites. Two aspects included in the development of the social conflict index were agriculture and fishing areas.	A ñcostò layer was developed to be incorporated into Marxan. The cost layer was constructed to avoid threats (see threats) and minimize conflict with fisheries and other uses (e.g., fishing density, oil and transport routes).	No costs were incorporated	A ñcost" layer was developed to be included in DSS analyses to avoid/minimize social conflict with artisanal fisheries. Artisanal (small) boat fishing pressure (mean standardized value per planning unit as density boats/km^2) was used as an additional variable to minimize in the selection of marine reserves to avoid conflict.

	Decision support tools	A GIS-based collaborative software program (Consensus Mapper) and methodology developed by McGill University was used for expert-guided area delineation. The software allows exploration of spatial data, discussion of decision priorities, and mapping of ecologically significant regions. Individual maps are overlaid to show areas of overlap and consensus.	A software to facilitate group decision-making was employed (i.e., Ventana Corp./GroupSystems), Species database system	Geographic Information System, SNIB (species database system)	Data integration and analysis, as well as delineation of conservation areas was supported by WWF-CECARENA DataMap2 tool, a GIS-linked software specialized in collecting spatial information for conservation targets. DataMap2 is a user-friendly software and was provided to experts before workshop to begin the compilation of data. Data integration and analysis, was supported by ArcView (ESRI 1998) Areas were selected by broad consensus (highly participative process), with active participation of independent experts and relevant stakeholders.	SPOT: Optimization software used to generate planning units and to automatically calculate the limits of each hexagonal unit to facilitate boundary length of portfolio (i.e., total perimeter of selected areas). Marxan: A decision support tool that helps to design conservation area (e.g., reserves) systems to achieve some minimum representation of biodiversity features (user-defined targets) for the smallest possible cost.	Geographic Information System, qualitative multicriteria analysis model (implemented in Excel)	SITES DSS facilitate the selection and design of representative systems of reserves for the conservation of biodiversity. It is a streamlined derivative of SPEXAN. An Arc-View extension in Avenue (VKMax.AVX) was created to evaluate targets other than habitats or individual species (e.g., fish species richness, fishing pressure). This was used to evaluate SITES portfolios (i.e., model solutions) that fulfil all targets and optimize different variables in potential marine reserve portfolios (e.g., evaluate whether conservation areas have an adjacent area located within a given threshold distance to maximise connectivity).
	Planning units	No planning units were used. Areas were manually delineated with the software based on expert knowledge, using a GIS interface.	No planning units were used. Areas were manually delineated by experts and later digitized.	No planning units were used. The planning team and workshop participants considered that spatial data (e.g., occurrences or distribution maps of species and habitats) was significantly biased and fragmented across the study region (i.e., Mexicoõs EEZ), and thematic cartography was not available at an adequate scale/resolution. Polygons were delineated by experts aided by GIS.	No planning units were used. Areas were manually delineated by experts, aided by a GIS-based software	A grid of uniform planning units (3500-hectare hexagons) was used. The same type and size of unit was used in another marine planning exercise using similar methodology (Southern California Marine Ecoregional Assessment). Size was considered adequate to represent relatively small ecological systems (e.g., coral reefs), without increasing the total number of analysis units, which was limited by computation capacity. On the other side, the hexagonal shape approximate a circle (therefore reducing perimeter/area ratio) and provide more borders to facilitate construction of more compact priority sites with adjacent units.	No planning units were used. Areas were manually delineated by experts, aided by a GIS-based software; national priority regions (CONABIO 1996) served as a reference and were modified and further refined.	Individual irregular planning units were islands, entire archipelagos covering a linear distance of 50km, sections of coast 50km in length, and individual mangroves with a buffer distance of 5 km on each side. These were selected as ecologically functional and implementable units. A conservative distance of 5km from shore was selected to include all coastal habitats.
	Conservation alternatives	Ecologically Significant Regions (ESRs) served to highlight places of high ecological significance. Priority Conservation Areas (PCA) are a subset of ESRs that have become priorities based on significant threats and/or opportunities. PCAs are not intended as an MPA or marine reserves network design, but rather as a portfolio of continentally significant sites which can serve as nodes around which a network of reserves can be built. It is a first step towards a continental conservation strategy for B2B species and ecosystems. Interface areas (e.g., coastal wetlands) and islands were included.	Marine Priority Regions (RMP): Of the 70 priority areas identified, 43 are located in the Pacific Ocean and 27 in the Gulf of Mexico and Caribbean Sea. 58 were classified (based on conglomerate analysis) as high biodiversity areas, 41 as areas that presented some type of threat, 38 correspond to areas important for natural resource use (by different sectors), and 8 are known to be biologically important, but more research is needed. For 3 areas available information at that time was not enough to classify them within any of the previous classes, but nevertheless were selected by experts. General recommendations about different protection and management alternatives are provided.	Marine Priority Sites (SMP): SMPs should be used to guide not only MPA network development, but other strategies (e.g., ocean zoning, integrated coastal management, impact assessment) should be considered, and refinement and more detailed regional or local planning exercises considering threats and biological diversity criteria should be developed to implement local actions. SMPs include important areas for threatened, protected, endemic and micro-endemic species, critical habitat, important wetlands, and area important for the development, feeding and reproduction of species, among others. In addition to the priority sites and gap analysis, a comprehensive biodiversity database (and cartography) with information on insular bodies was constructed. Includes coastal and deep sea sites. No zoning was considered.	Priority conservation areas were defined as Integrated Biologically Important Areas (AIBI) for the conservation of coastal-marine biodiversity (6,118,530 ha). The 24 sites (and associated islands) represent the areas of overlap between areas of importance for species, habitats, oceanographic processes and ecological processes, for marine, coastal and terrestrial areas. Important coastal lagoons sites include the surrounding land and a marine buffer of 30 km from the coastline. In addition, maps of potential social conflict and anthropogenic threats (current and projected) were developed. AIBI areas, threat and conflict maps were not integrated to select or prioritize areas for conservation action. Results show that the areas subject to high anthropogenic pressure coincide with the areas of biological importance. One expected use of the results is to help the stakeholders develop a meaningful conservation portfolio.	Priority Conservation Sites (portfolio): The portfolio aim was to serve as a reference for important coastal and marine biodiversity areas. It was intended to guide a diversity of conservation actions, from the implementation of marine protected areas (e.g., marine reserves) to natural resource (e.g., fisheries) management. The exercise included selection of pelagic sites not considered by previous prioritization exercises. The sites selected by Marxan based on the benthic model were rejected by experts.	Priority areas for the conservation of coastal and marine resources (45) and terrestrial priority regions (14). Areas are prioritized based on weighted ecological, social, economic and cultural criteria. Priority areas include: coastline portions; islands and archipelagos; and marine and coastal areas (coastal lagoons, estuaries, wetlands and marine areas). Associated to priority areas are regional actions directed to improve marine resource management and conservation. Actions relate to fisheries management, ecotourism/use of islands, coastal zone use, and protected areas; but not all actions are spatial (e.g., improve regulations, management decentralisation, information access-sharing, capacity building, outreach). Actions are associated with scale of intervention, time frame and particular institutions or stakeholders. Spatial actions included: land/marine spatial planning, creation of reserves, socioeconomic and ecological research/inventories.	Coastal Marine Reserves for Rocky Fish and Habitat Conservation: Sites represent a network of potential marine reserves (i.e., marine protected areas). Two scenarios were explored: biological optimal network (no costs) and network that reduces social conflict by excluding areas where fishing pressure and conservation collide. Maximum distance between adjacent reserves is 89 km (mean, 40 km) ensuring connectivity for most fishes and many invertebrates. Reserves are large enough to ensure local retention of >90% algae propagules and >45% fish and invertebrate larvae. Minimizing conflict did not significantly decrease achievement of targets, e.g., coral reefs (87% vs. 100% target) and spawning aggregations (90% vs. 100% target).
Synthesis	Scheduled areas and actions	Not considered	Not considered	Sites were classified as important, very important and extremely important based on expert opinion (not necessarily indicating scheduling of conservation actions). Criteria???	Not considered	Scheduling has been considered in later phases for implementation of the portfolio.	Areas and actions are prioritised based on biological, socioeconomic, opportunities and cultural criteria. Both are presented in a scale of relative importance, and actions are associated to a general timeframe.	Not considered, the portfolio represents an integral reserve network to achieve the defined conservation objectives.
	Mainstream technical outputs	Brochure, workshops, meetings and conferences, addressed to federal agencies (main recipients of outputs). Information and datasets are also available through the CEC website or on request. No specific products appropriate for different stakeholders (in charge of implementing or affected, interested) were produced.	Book with summary of methods and description of priority areas, as well as website, for the use of decision-makers, government and conservation practitioners; not for other relevant stakeholders	Brochure with summary for government and international organizations prepared, not for other relevant stakeholders.	A couple of CDs with all the compiled information and the resulting maps of the workshop were delivered to all participants in the workshop and a scientific paper reviewed the process (Enriquez-Andrade et al. 2005). No brochure, technical report or specific outreach products for different stakeholders (e.g.., agencies, managers, fishermen, general public) were produced. A database (Prometeo) was constructed with the intention of updating and giving access to all participants and other stakeholders.	A technical report was produced, accompanied by datasets and metadata database. TNC provided the analysis and portfolio for a national workshop that analysed the gaps in marine conservation and defined priorities at a national scale (CONABIO 2007). No specific products (e.g., brochures) for particular stakeholders (e.g., managers, fishing federations, general public) were developed.	A technical report was produced and made available to stakeholders and workshops participants.	Only peer-reviewed paper (available to scientists)
	Apply conservation actions	Due to financial restrictions follow up on the planning process was not possible, but the results have been used by governments and the CEC to continue developing the tri-national strategy of the North America MPA Network, which also considers the west coast. The constraint of top-down efforts highlights the necessity of eventually matching this project with a community-based action plan involving members of the communities within the PCAs.	The results of the planning exercise have been used by subsequent planning exercises, as this exercise is meant to serve as a reference framework for decision-makers, scientists, managers and general public that participate in planning for conservation, resource use, management and scientific research, as well as in the allocation of financial resources.	Not yet evaluated. The project is not considered as a final and static product and should be refined and updated based on new information, but also used as a reference for more detailed and regional planning and conservation projects.	The results of the planning exercise have been used by subsequent planning exercises, some of which have been partially implemented or are in the process of implementation, though not in a systematic or coordinated way. Some of the planning outputs have been used as a basis or support for conservation actions, such as the decree of Marine Protected Areas (e.g. Bah²a de los Angeles-Canales de Ballenas y Salsipuedes Biosphere Reserve).	The portfolio has been used by The Nature Conservancy and World Wildlife Fund as a framework for their conservation activities in the region and strategy to use the portfolio as the design for implementation of a network of conservation and managed areas for the Gulf of California. Follow up studies have been using it as a reference (e.g., studies in biological and social connectivity between priority sites). It was an important (scientifically robust) input for the National marine gap analysis (CONABIO 2007). Some protected/managed areas have been decreed since its completion, and this and other exercises have been cited as a reference to focus attention (e.g., Infiernillo Channel RAMSAR site, Guaymas Basin and Eastern Pacific Rise Hydrothermal Vents Sanctuary; TNC and CIAD provided technical support and data to Mexicoõs National Protected Areas Commission).	The portfolio has been used by Pronatura as a framework for their conservation activities in the region.	Not developed for implementation. The main goal of the exercises was to develop a method/model for marine reserve selection.
	Monitoring	No monitoring strategy. Mention to the need of monitoring achievement of conservation objectives.	No monitoring strategy	Not monitoring strategy. Mention to the need of monitoring achievement of conservation objectives.	No monitoring strategy	There are monitoring strategies developed for the collaboration between TNC and WWF and the network of conservation and managed areas, but this has happened after the ecoregional assessment was completed.	No monitoring strategy	No monitoring strategy
NOTES		PCAs were refined to reflect the goals of the CEC process, the variable nature of data available in the three nations and the spatial scale of the Baja California to the Bering Sea (B2B) region.	The process not only identified important areas for marine biodiversity conservation, but also those that are relevant in terms of natural resource use (some of these are coincident with high biodiversity areas) or of which there was few information and further biodiversity research is needed; aims to serve as a reference framework for decision-makers, scientists, managers and general public that participate in planning for conservation, resource use, management and scientific research, as well as in the allocation of financial resources.	The project uses results from previous national and regional planning exercises (CONABIO 1998, Ulloa 2006, Morgan et al. 2005, Enriquez et al. 2005) and a survey to experts, academics and conservation practitioners (Encuesta Nacional, CONANP-CONABIO-Pronatura-TNC-FMCN-INE 2006) in the process of identifying marine priority conservation areas and developing a national marine gap analysis.	The whole planning exercise, including the compilation of information and data preparation for the workshop, as well as the stakeholder engagement process lasted around three years.	The project builds on previous exercises and uses both compiled datasets and priority sites portfolios (i.e. Morgan et al. 2005, Enriquez et al. 2005, conservation proposal (map) for the GoC Marine Planning Process: INE-OEMGC) to produce a new and updated plan, based on TNC's ERAs methodology. In 2005, experts reviewed the portfolio and proposed additions and changes, which will be incorporated in a later planning phase for strategy development.	The planning process focused on promoting biodiversity conservation by identifying and prioritising conflicts or problems perceived by stakeholders interested in particular ecosystems or species (e.g., charismatic species), resources (e.g., commercial species) or natural areas (protected or not). Results provides the basis for strategic planning for actions directed to medium- and long-term conservation, management, and sustainable use of marine/coastal resources. Regional priority areas were identified using as a reference the national analysis undertaken by CONABIO (1996).	The project is focused in developing a model to design reserve design and is only focused on reef fishes and rocky shores habitats. Therefore important elements of biodiversity, such as marine mammals, seabirds and sea turtles, pelagic components and oceanographic processes are not well covered by the proposed reserve network. The data demand of the model, implies a high sampling effort and specialized, but not necessarily financially unviable for other planning. As the study suggests, spatial conservation demands of fish and associated habitats could partially cover other conservation objects not directly targeted by the planning process.


Baja to Bering - B2B
Priority Conservation Areas (PCA)
Regiones Marinas Prioritarias
Marine Priority Regions (RMP)
Pacific Ocean
National Marine Gap Analysis
Marine Priority Sites (SPM)
Coalicion - CSGC
Integrated Biol. Important Areas (AIBIs)
Eco-regional Assessment - ERA
Priority Conservation Sites
Marine Reserve Network - MRN
Coastal Marine Reserves
Protected areas
Cortezian Ecoregion
Km 0 100 200 300 400

Perceptions of achievement of planning goals, implementation, and benefits

Planning exercise: [ ]

1. Mark what were the **goals** [1] of the planning exercise, include reported (explicitly stated) and implicit (not reported or emerging after the original goals were formulated); check all that apply.

	REPORTED	IMPLICIT
Identify generic priority areas (no particular actions specified) for conservation actions[2]	☐	☐
Identify priority areas associated with specific conservation actions	☐	☐
Guide further conservation/resource use planning	☐	☐
Document and assess biodiversity	☐	☐
Document and assess problems/threats/conflicts	☐	☐
Document and assess opportunities for or constraints on conservation management	☐	☐
Evaluate protected area system (gap analysis)	☐	☐
Influence on where and how leading/partner organisation(s) should focus their efforts	☐	☐
Other:	☐	☐

2. Overall, the reported planning goals have been achieved.

   ☐ Strongly disagree ï Not at all
   ☐ Disagree ï Poorly
   ☐ Neither agree nor disagree ï Moderately
   ☐ Agree ï Satisfactorily
   ☐ Strongly agree ï Fully

   Why or why not?

   [ ]

3. Overall, implicit planning goals have been achieved.

   ☐ Strongly disagree ï Not at all
   ☐ Disagree ï Poorly
   ☐ Neither agree nor disagree ï Moderately
   ☐ Agree ï Satisfactorily
   ☐ Strongly agree ï Fully

   Why or why not?

   [ ]

4. If the plan has influenced or led to spatial implementation of actions [3], what were they? (check all that apply)

   ☐ Establish multiple-use protected areas
   ☐ Establish no-take areas
   ☐ Develop/update coastal-marine zoning plans: fisheries [4]
   ☐ Develop/update coastal-marine zoning plans: others [5]
   ☐ Develop/update coastal zone development plans
   ☐ Socioeconomic/ecological research/monitoring
   ☐ Surveillance (e.g., *vs*. illegal fishing)
   ☐ Restoration/population restocking
   ☐ Control/eradication of exotic invasive species
   ☐ Other: _____

[1] We refer to goals related to identifying spatial priorities.
[2] Actions relate to no-take areas, multiple use zones, fisheries management/controls, restoration, etc.
[3] Includes actions in progress or that have been scheduled.
[4] Includes spatial allocation of gear/species restrictions, bag/size limits, temporal closures, etc.
[5] Includes maritime routes, industrial activities (e.g., oiling), etc.

If you have some examples of the actions marked above, please include them here.

5. What are the main factors[6] that have **enabled/facilitated** the spatial implementation of actions**:**

1. 2. 3.

6. What are the main factors that have **constrained/limited** the spatial implementation of actions**:**

1. 2. 3.

7. Aside from spatial implementation of actions, did planning help to (if yes, was it unanticipated?):

Financial benefits	**YES**	**Unanticipated**	**NO**
Make better use of available conservation resources	☐	☐	☐
Create/maintain funding leverage to access more conservation resources	☐	☐	☐
Improve efficiency in the operations of organisations/agencies	☐	☐	☐
Social benefits	**YES**	**Unanticipated**	**NO**
Create a regional conservation approach/shared vision for conservation	☐	☐	☐
Improve collaboration/coordination between organisations/agencies	☐	☐	☐
Bring stakeholders together (that otherwise would not interact)	☐	☐	☐
Facilitate access and sharing of socioecological data between organisations/agencies	☐	☐	☐
Build trust among organisations/agencies	☐	☐	☐
Attitude benefits	**YES**	**Unanticipated**	**NO**
Raise awareness of biodiversity values and environmental problems	☐	☐	☐
Promote a positive attitude towards marine conservation	☐	☐	☐
Encourage stakeholders commitment to conservation actions	☐	☐	☐
Knowledge and skills benefits	**YES**	**Unanticipated**	**NO**
Synthesise existing data on ecological, social and cultural values	☐	☐	☐
Collect new knowledge on ecological or social values	☐	☐	☐
Build supporting scientific evidence for decision-making by organisations/agencies/partners	☐	☐	☐
Gain knowledge of new tools and concepts	☐	☐	☐
Improve conservation planning methods and theory (academic)	☐	☐	☐
Develop skills/expertise of organisations/agencies in conservation planning	☐	☐	☐
Apply personal learning in later planning exercises	☐	☐	☐
Apply institutional learning in later planning exercises	☐	☐	☐
Transfer conservation planning skills and knowledge to local communities/organisations	☐	☐	☐
Institutional benefits	**YES**	**Unanticipated**	**NO**
Standardise and/or have more rigorous planning processes	☐	☐	☐
Validate ongoing conservation actions/programmes	☐	☐	☐
Identify a portfolio of priority areas used by organisations/agencies/partners	☐	☐	☐
Meet regional/national/international commitments	☐	☐	☐

[6] Factors may refer to characteristics of the planning process and outputs, but also to the socioeconomic context in which implementation occurs.

Influence conservation planning/strategies/actions by organisations/agencies/partners	☐	☐	☐
Facilitate the development/implementation of regional or local conservation strategies	☐	☐	☐
Create/maintain policy leverage to improve policies, laws, norms, codes, etc.	☐	☐	☐
Other:	☐	☐	☐

8. Has there been monitoring/assessment on the outcomes/effectiveness of implementing spatial actions of the plan?

☐ Yes ☐ No

9. Has the plan been updated or are there intentions to update it?

☐ Yes ☐ No

10. What are the main **benefits** of having multiple conservation plans for the Gulf of California?

11. What are the main **difficulties** of having multiple conservation plans for the Gulf of California?

Supplementary Material C4: Catchment model parameterisation and data preparation

1. Catchment delineation

Digital elevation model

We used a 240m resolution hydrologically-corrected and pit-filled digital elevation model (DEM) to delineate streams. Accuracy in the delineation of the stream network was an important consideration in our study, since this process is particularly sensitive to flat areas, such as the coastal plains of the western coast of the Gulf of California. We used ArcHydro Tools[1] (ESRI 2009) to create a hydrologically-corrected and conditioned DEM that better represented the stream network. For this reason we decided to burn-in vector hydrography (CONABIO 2008) into the DEM following the AGREE method proposed by Hellweger (1997). However, in order to construct the floodplain grid (discussed below) we used an unmodified DEM. The Pit-Filler tool (provided with SedNet) was also used to remove artificial sinks from the DEM. The catchment delineation procedure in N-SPECT automatically uses the Fill command in ArcGIS to eliminate sinks.

Catchment delineation

Main considerations to define subcatchments size were: size of the planning domain (larger extents usually require larger analysis/planning units); subcatchments area (we aimed to minimise variance subcatchments size, important when using area as a proxy for catchment management costs because larger subcatchments may be disproportionately more costly), resolution of other datasets (1:250,000 land cover and soil datasets were our main constraints); uncertainty in load estimates (larger catchments hold smaller uncertainty in the catchment model); and suitability for other purposes. For the latter, we intended on using subcatchments as planning units in prioritisation analysis using the conservation planning software Marxan. The number of analysis units strongly influence processing time; and shape uniformity can influence prioritisation results when clumping is used in Marxan.

During the N-SPECT automated subcatchment delineation, the maximum value from the flow accumulation grid is multiplied by one of several predefined threshold values (0.03, 0.06, 0.1) to create small, medium or large subcatchments. This means that a small subcatchment is 3% of the maximum flow accumulation value, a medium subcatchment is 6% of the maximum flow accumulation value, and a large subcatchment is 10% of the maximum flow accumulation value. We used the 'small' subcatchment option, which resulted in 27 subcatchments with mean size of 3774km^2 (385 to 8652km^2). For SedNet, we compared different subcatchments delineations based on permutations of drainage area thresholds – DAT (40, 50 and 60 km^2) and minimum first-order stream

[1] ArcHydro functionality within ArcGIS is used to delineate watershed areas and calculate the flow direction and accumulation grids.

length threshold – SLT (1 and 2km). The combination we used, based on considerations such as subcatchment size and resolution of datasets, was DAT = 40km^2 and SLT = 1km, which resulted in 1,655 subcatchments, with a mean size of ~60 km^2 (STD = 39 km^2). An additional set of stream network parameters are: channel width coefficient, channel width area exponent and channel width slope exponent. We used default values (6.04, 0.26 and 0, respectively) because we did not have information to justify using different values.

An inevitable problem when defining catchments using DAT > 40km^2 is that many small coastal catchments (where land use is dominated by urban and cropland and can be significant sources of pollutants) were not delineated. The selected parameters still resulted in a few small coastal catchments not being delineated. An analysis of the potential contributions of pollutants from these small coastal catchment revealed that their degree of influence was not important, and therefore they were excluded from subsequent analyses. If deemed necessary (e.g., under different circumstances or in a fine-grained analysis), this limitation could be addressed by generating a mask of those areas and multiply the runoff raster by RUSLE parameters and nutrient concentration rasters to calculate the additional loads. This problem is handled by N-SPECT during catchment delineation, thus small coastal catchments are also incorporated into subsequent analysis.

2. Land cover/vegetation

Landcover classes

The selected landcover datasets (SEMARNAT-UNAM 2002) have been used in previous erosion and landcover change studies (SEMARNAT-UACH 2002; Velázquez et al. 2002). Lower level classification levels (i.e., vegetation type and community) was required to classify types grouped under “other cover/vegetation types” at the formation level (see **Table 1**). Errors in the assignment of classes at the formation level are relatively small in comparison with the higher uncertainty in lower classes (e.g., sub-formation and vegetation type) and for these reasons it is not recommended to use lower classes for land cover change models at this scale (Velázquez et al. 2002).

Vegetation cover

The cover factor reduces the soil loss estimate according to the effectiveness of vegetation at preventing detachment and transport of soil particles. Values vary between 0 and 1, with higher values representing low cover, i.e., areas that are more prone to erosion. We decided not to use the C values tabulated in SEMARNAT-UACH (2002) for the following reasons: the values provided did not include all the landcover classes defined for our study, there were differences between the classes (even within the same study (e.g. scrub/brush classes); the map was only available for INEGI Series II

(1993), and to keep it as close as possible to the values used in other regional studies (e.g., Mesoamerica: Burke & Sugg 2006); if consistently used it can be useful for comparing areas.

Table 1. Landcover formation classes and corresponding vegetation types and communities

ID	Landcover class	Formation	Sub-formation	Vegetation type	Community
1	Urban	Otras coberturas	**Asentamiento humano**	Asentamiento humano	Asentamiento humano
2 - 6	Cropland	**Cultivos**	Cultivos	Agricultura de riego y humedad Agricultura de temporal Plantación forestal	Agricultura de riego Agricultura de humedad Riego suspendido Agricultura de temporal
7	Pasture	**Pastizales inducidos y cultivados**	Pastizales inducidos y cultivados	Pastizales inducidos y cultivados	Pastizal inducido Pastizal cultivado Sabana
8	Grassland	**Pastizal natural**	Pastizal natural	Pastizal natural	Pastizal natural Pradera de alta montana
9	Woodland	**Bosques**	Bosques Bosques secundarios	Bosque de coníferas Bosque de coníferas y latifoliadas Bosque de latifoliadas Bosque mesofilo de montana	Bosque de pino Bosque de pino-encino Bosque de oyamel Bosque de encino Bosque bajo-abierto Bosque de tascate Bosque mesofilo de montana
10	Rain forest	**Selvas**	Selvas Selvas secundarias	Selva caducifolia y subcaducifolia Selva perennifolia y subperennifolia	Matorral subtropical Selva baja subperennifolia Selva baja caducifolia y subcaducifolia Selva baja espinosa Selva mediana caducifolia y subcaducifolia
11	Dry forest	Selvas	Selvas Selvas secundarias	Selva caducifolia y subcaducifolia	**Matorral subtropical** **Selva baja espinosa** **Chaparral (Matorral)**
12	Palm forest	Otros tipos de vegetación	Otras coberturas de vegetación	Otros tipos de vegetación	**Palmar**
13	Scrub/Shrub	**Matorral**	Matorral	Matorral xerófilo Mezquital	Matorral desértico rosetofilo Matorral desértico microfilo Matorral rosetofilo costero Matorral sarcocaule Matorral crasicaule Matorral sarcocrasicaule Matorral sarcocrasicaule de neblina Matorral submontano Mezquital
14	Halophytic	Otros tipos de vegetación	Otras coberturas de vegetación	Otros tipos de vegetación	**Vegetación halófila y gipsofila**
15	Wetland	**Vegetación hidrófila**	Otras coberturas de vegetación	Vegetación hidrófila	Vegetación de galería Popal-tular Manglar
16	Dunes	Otros tipos de vegetación	Otras coberturas de vegetación	Otros tipos de vegetación	**Vegetación de dunas costeras** **Vegetación de desiertos arenosos (matorral)**
17	Bare land	Otros tipos de vegetación	Otras coberturas de vegetación	**Área sin vegetación aparente**	Área sin vegetación aparente
18	Water	Otras coberturas	**Cuerpo de agua**	Cuerpo de agua	Cuerpo de agua

Green text indicates the classification level used to define the landcover class.

Orange text indicates communities that were re-classified under a different formation (for the purpose of this study) based on hydrological characteristics.

3. Riparian vegetation

Due to technical requirements/costs, riparian vegetation has been traditionally mapped at fine scales for local studies based on on-site and remote sensed (mostly aerial photography) data (Congalton et al. 2002; Etienne 1997). Thus, spatial data is commonly not available or consistent for regional or national scale studies. Differences in mapping accuracy were evident in class mismatches between different periods of our land use time series, e.g., areas classified as woodland was also classified as dry forest and scrub/shrub in subsequent datasets.

Due to inconsistencies between landcover datasets for the "riparian vegetation" class we decided to create a new map for this class. First, we combined the polygons identified as riparian vegetation (i.e., *bosque de galeria*) from each of the 3 periods (1976, 1993, and 2000) of the time series. Then, we buffered (80m) the delineated stream network (see above) and used it to clip polygons corresponding to natural tree-cover vegetation classes (i.e., *bosque*, *selva*, *palmar*, *hidrofila*). Finally, we combined both layers into a single layer that represents the maximum extent of current (2000) riparian vegetation for the study region. For the purpose of modelling sediment trapping, this approach to map riparian vegetation is appropriate (Wilkinson et al. 2008), but improved maps and standardised methods to map riparian vegetation at regional scales for future studies are needed.

4. Soils

Hydrologic group

There are four HSG groups ranging from soils with high infiltration capacity (A), e.g., high sand content, to soils characterised by a very slow rate of water transmission (D), e.g., with high clay content (**Table 2**). For Mexico CNA (1987) proposes a corresponding HSG value based on FAO soil types (**Table 3**).

Table 2. Classification of soils types in terms of HSG.

HSG	Soil Texture	Soil Group Characteristics
A	Sand, loamy sand, or sandy loam	Soils having high infiltration rates, even when thoroughly wetted and consisting chiefly of deep, well-drained to excessively drained sands or gravels. These soils have a high rate of water transmission.
B	Silt loam or loam	Soils having moderate infiltration rates when thoroughly wetted and consisting chiefly of moderately deep to deep, and moderately fine to moderately coarse textures. These soils havea moderate rate of water transmission.
C	Sandy clay loam	Soils having slow infiltration rates when thoroughly wetted and consisting chiefly of soils with a layer that impedes downward movement of water, or soils with moderately fine to fine texture. These soils have a slow rate of water transmission.
D	Clay loam, silty clay loam, sandy clay, silty clay, or clay	Soils having very slow infiltration rates when thoroughly wetted and consisting chiefly of clay soils with a high swelling potential, soils with a permanent high water table, soils with a claypan or clay layer at or near the surface, and shallow soils over nearly impervious material. These soils have a very slow rate of water transmission.

Table 3. HSG values based on soil types found in Mexico

KEY	UNIT	HSG	KEY	UNIT	HSG	KEY	UNIT	HSG
Ao	Acrisoles órticos	C	Fh	Ferrasoles húmicos	D	We	Planosoles éutricos	D
Af	Acrisoles férricos	D	Fa	Ferrasoles acrícos	D	Wd	Planosoles dístricos	D
Ah	Acrisoles húmicos	C	Fp	Ferrasoles plínticos	D	Wm	Planosoles mólicos	D
Ap	Acrisoles plínticos	C	Je	Fluvisoles eútricos	B	Wh	Planosoles húmicos	D
Ag	Acrisoles gléicos	D	Jc	Fluvisoles calcáreos	B	Ws	Planosoles solódicos	D
To	Andasoles ócricos	A-B	Jd	Fluvisoles dístricos	B	Wx	Planosoles gélicos	D
Tm	Andasoles móticos	A-B	Jt	Fluvisoles tiónicos	B	Dg	Podzoluvisol gléico	B
Th	Andasoles húmicos	A-B	Jg	Fluvisoles gléyicos	C	Dd	Podzoluvisol dístrico	B
Tv	Andasoles vítricos	A-B	Ge	Gleysoles eútricos	D	De	Podzoluvisol eútrico	B
Qf	Arenosoles ferráticos	A-B	Gc	Gleysoles calcáreos	D	LL	Ranker	C
Ql	Arenosoles lúvicos	A-B	Gd	Gleysoles dístricos	D	Re	Regosoles eútricos	B
Qc	Arenosoles cámbicos	B	Gm	Gleysoles mólicos	D	Rc	Regosoles calcáreos	B
Qa	Arenosoles álbicos	A-B	Gh	Gleysoles húmicos	D	Rd	Regosoles dístricos	B
Bd	Cambisoles dístricos	D	Gp	Gleysoles plínticos	D	Rx	Regosoles gélicos	D
Be	Cambisoles eútricos	C	Oe	Hístosoles eútricos	D	E	Rendzinas	C
Bh	Cambisoles húmicos	C	Od	Hístosoles dístricos	D	Zo	Solochaks órticos	D
Bg	Cambisoles gléicos	D	Ox	Hístosoles gélicos	D	Zm	Solochaks mólicos	D
Bx	Cambisoles gélicos	D	I	Litosoles	D	Zt	Solochaks taquíricos	D
Bk	Cambisoles cálcicos	C	Lo	Luvisoles órticos	C-D	Zg	Solochaks gléicos	D
Bc	Cambisoles crómicos	C	Lc	Luvisoles crómicos	C-D	So	Solonetz órticos	D
Bv	Cambisoles vérticos	D	Lk	Luvisoles cálcicos	C-D	Sm	Solonetz mólicos	D
Bf	Cambisoles ferrálicos	C	Lv	Luvisoles vérticos	C-D	Sg	Solonetz gléicos	D
Kh	Kastanozems háplicos	C-D	Lf	Luvisoles férricos	C-D	Sa	Solonetz álbico	D
Kk	Kastanozems cálcicos	C-D	La	Luvisoles álbicos	C-D	Vp	Vertisoles pélicos	C-D
KI	Kastanozems lúvicos	C-D	Lp	Luvisoles plínticos	C-D	Vc	Vertisoles crómicos	C-D
Cl	Chermozen lúvico	C	Lg	Luvisoles gléicos	C-D	Xh	Xerosoles háplicos	B-C
Ck	Chermozen cálcico	B	Ne	Nitosoles eútricos	C	Xk	Xerosoles cálcicos	B-C
Ch	Chermozen háplico	B	Nd	Nitosoles dístricos	C	Xo	Xerosoles gípsicos	B-C
Hh	Phaeozems háplicos	C	Nh	Nitosoles húmicos	C	Xl	Xerosoles lúvicos	D
Hc	Phaeozems calcáreos	C	Po	Podzoles órticos	A-B	Yh	Yermosoles háplicos	C
Hl	Phaeozems lúvicos	C	Ph	Podzoles húmicos	A-B	Yk	Yermosoles cálcicos	C
Hg	Phaeozems gléicos	D	Pp	Podzoles plácicos	A-B	Yg	Yermosoles gípsicos	C
Fo	Ferrasoles órticos	D	Pg	Podzoles gléicos	A-B	Yl	Yermosoles lúvicos	D
Fx	Ferrasoles xánticos	D	Pp	Podzoles plácicos	A-B	Yt	Yermosoles taquirícos	D
Fr	Ferrasoles ródicos	D	Pg	Podzoles gléicos	A-B			

Erodibility

SEMARNAT-UACH (2002) compared the 3 methods (Wischmeier and Mannering equation; Wischmeier Nomogram, based on laboratory work with 134 soil samples covering all soil types in all climatic regions; and FAO method using INEGI soil types and texture maps), the study found significant differences in the values and concluded none is accurate or performs better than the others but decided using FAO while better estimations are developed. We calculated erodibility using K-factor values summarised by SEMARNAT-UACH (2002), based on the method proposed by FAO (1980); see **Table 4**.

For soils with multiple horizons, CNA (1991) suggests a *weighted sum of K-factors*, assuming the first soil horizon (S_1) represents 60% and the second (S_2) 40% (i.e., K = S_1K-value *0.6 + S_2K-value *0.4); if 3 horizons[2] are represented in the soil the assumed percentages are 50%, 30% and 20%, respectively (i.e., K = S_1K-value*0.5 + S_2K-value*0.3 + S_2K-value*0.2). When a polygon is over water, units do not contain k-factor or hydrologic soil group data and thus were assigned a value of zero (and HSG: D); other examples of this situation also occur with urban soils, in this case a K value of 0.30 was used.

[2] INEGI 1:1,000,000 map was used; the layer contains data for 3 soil horizons, which allows to calculate a more precise K (weighted) value

Table 4. Soil erodibility (K-factor) corresponding to each soil type-texture combination

Soil type	Coarse	Medium	Fine	Soil type	Coarse	Medium	Fine
A	0.026	0.04	0.013	*Lo*	0.026	0.04	0.013
Af	0.013	0.02	0.007	*Lp*	0.053	0.079	0.026
Ag	0.026	0.04	0.013	*Lv*	0.053	0.079	0.026
Ah	0.013	0.02	0.007	*M (a,g)*	0.026	0.04	0.013
Ao	0.026	0.04	0.013	*N (d.e.h)*	0.013	0.02	0.007
Ap	0.053	0.079	0.026	*O (d,e,x)*	0.013	0.02	0.007
B	0.026	0.04	0.013	*P*	0.053	0.079	0.026
Bc	0.026	0.04	0.013	*Pf*	0.053	0.079	0.026
Bd	0.026	0.04	0.013	*Pg*	0.053	0.079	0.026
Be	0.026	0.04	0.013	*Ph*	0.026	0.04	0.013
Bf	0.013	0.02	0.007	*PI*	0.026	0.04	0.013
Bg	0.026	0.04	0.013	*Po*	0.053	0.079	0.026
Bh	0.013	0.02	0.007	*Pp*	0.053	0.079	0.026
BK	0.026	0.04	0.013	*Q (a,c,f,l)*	0.013	0.02	0.007
Bv	0.053	0.079	0.026	*R*	0.026	0.04	0.013
Bx	0.053	0.079	0.026	*Re*	0.026	0.04	0.013
C (g, h, k, l)	0.013	0.02	0.007	*Rc*	0.013	0.02	0.007
D (d, e, g)	0.053	0.079	0.026	*Rd*	0.026	0.04	0.013
E	0.013	0.02	0.007	*Rx*	0.053	0.079	0.026
F (a,h,o,p,r,x)	0.013	0.02	0.007	*S*	0.053	0.079	0.026
G	0.026	0.04	0.013	*Sg*	0.053	0.079	0.026
Gc	0.013	0.02	0.007	*Sm*	0.026	0.04	0.013
Gd	0.026	0.04	0.013	*So*	0.053	0.079	0.026
Ge	0.026	0.04	0.013	*T*	0.026	0.04	0.013
Gh	0.013	0.02	0.007	*Th*	0.013	0.02	0.007
Gm	0.013	0.02	0.007	*Tm*	0.013	0.02	0.007
GP	0.053	0.079	0.026	*To*	0.026	0.04	0.013
Gx	0.053	0.079	0.026	*Tv*	0.026	0.04	0.013
Gv	0.053	0.079	0.026	*U*	0.013	0.02	0.007
H (c,g,h,l)	0.013	0.02	0.007	*V (c,p)*	0.053	0.079	0.026
I	0.013	0.02	0.007	*W*	0.053	0.079	0.026
J	0.026	0.04	0.013	*Wd*	0.053	0.079	0.026
Jc	0.013	0.02	0.007	*We*	0.053	0.079	0.026
Jd	0.026	0.04	0.013	*Wh*	0.026	0.04	0.013
Je	0.026	0.04	0.013	*Wm*	0.026	0.04	0.013
Jt	0.053	0.079	0.026	*Ws*	0.053	0.079	0.026
Jp	0.053	0.079	0.026	*Wx*	0.053	0.079	0.026
K (h,K,l)	0.026	0.04	0.013	*X (h,k,l,y)*	0.053	0.079	0.026
L	0.026	0.04	0.013	*Y (h,k,l,y,t)*	0.053	0.079	0.026
La	0.053	0.079	0.026	*Z*	0.026	0.04	0.013
Lc	0.026	0.04	0.013	*Zg*	0.26	0.04	0.013
Lf	0.013	0.02	0.007	*Zm*	0.013	0.02	0.007
Lg	0.026	0.04	0.013	*Zo*	0.026	0.04	0.013
Lk	0.026	0.4	0.013	*Zt*	0.056	0.079	0.026

Curve numbers

We adjusted the CN value corresponding to each landcover class based on the standard SCS classification (C-CAP). Usually water is assigned with a 0 (zero) value, but some studies assign 100 instead (N-SPECT default is zero). For example, urban (high intensity development) class [A/89, B/92, C/94, D/95] have higher CN values than woods (good) class [A/30, B/55, C/70, D/77]. We considered that natural vegetation covers as having good hydrological condition, except low evergreen tropical forest (fair) and dry tropical forest (poor). CN are assigned within N-SPECT by overlaying the landcover and soil HSG grids (NOAA 2008). We used ArcCN-Runoff extension of ArcGIS (Zhan & Huang 2004) to independently determine CN and compare them with values used in other studies using the same methodology in catchment within our study region (Hernández-Guzmán et al. 2008).

Landcover crosswalk criteria and rationale

Urban.- Not possible to distinguish different development levels at the study scale; an average of the HSG/CN numbers for the 3 major components of urban areas was calculated (streets and roads, urban districts and residential districts); the average of high and medium-intensity developed was used (NOAA 2008, USDA 1986).

Cropland.- Not possible to distinguish different crop classes at the study scale; an average of the HSG/CN values for all cultivated land classes was calculated; cover factor was calculated as the average of C values [from a study on hydrological erosion in Mexico (SEMARNAT-UACH 2002)] of major crops existing in the study area, weighted to reflect their relative importance in terms of cultivated surface at regional scale. Note: If data is available the general (i.e., average) "crops/cultivos" value can be modified and adjusted to reflect the proportional surface that different crops occupy within the region or each study section (i.e., catchment/basin/catchment group).

Pasture (induced/culrivated grassland).- The HSG/CN values for pasture, grassland or range - continuous forage for grazing (fair condition) were used (USDA 1986); cover value was taken from a study on hydrological erosion in Mexico (SEMARNAT-UACH 2002).

Grassland (natural grassland).- The HSG/CN values for pasture, grassland or range - continuous forage for grazing (good condition) were used (USDA 1986); cover value was taken from default N-SPECT values (Note: The value for grasslands from the study on hydrological erosion in Mexico (SEMARNAT-UACH 2002) included urban green areas, so it was decided not to use that value).

Halophyte vegetation.- The HSG/CN values for herbaceous - mixture of grass, weeds, and low-growing brush, with brush the minor elements (fair) were used (USDA 1986); CN for "A" HSG type

was not available, preliminary this was calculated as the average of desert shrub. The C factor value used was the same as for natural grasslands (SEMARNAT-UACH 2002).

Woodland.- It was not possible to distinguish different woodland types (or vegetation cover levels) with confidence in available land cover time series; all wood classes were aggregated into woods (good condition) and are presented in a single class. The HSG/CN values for woods (good condition) were used (USDA 1986); cover value was calculated as the average for all corresponding wood types (i.e., oak, pine, conifers) from a study on hydrological erosion in Mexico (SEMARNAT-UACH 2002).

Tropical forest.- The HSG/CN values for woods (fair condition) were used (USDA 1986); cover value corresponds to low evergreen tropical forest (selva baja perenifolia) class derived from a study on hydrological erosion in Mexico (SEMARNAT-UACH 2002).

Dry tropical forest.- The HSG/CN values for woods (poor) were used (USDA 1986); cover value corresponds to low dry/thorn tropical forest (selva baja espinosa) class derived from a study on hydrological erosion in Mexico (SEMARNAT-UACH 2002). This vegetation type could be more similar to some types of the scrub/shrub that to tropical forest.

Scrub/shrub.- The HSG/CN values for desert shrub - major plants in class include saltbush, greasewood, creosotebush, blackbrush, bursage, palo verde, mesquite, and cactus - (good) were used (USDA 1986); cover value was calculated as the average for all desert scrub/shrub types from a study on hydrological erosion in Mexico (SEMARNAT-UACH 2002).

Palm vegetation.- The HSG/CN values for brush (good) was used; cover-factor was maintained (NOAA 2008, USDA 1986).

Wetland.- The HSG/CN and cover factor values for Estuarine Forested Wetland were used (NOAA 2008, USDA 1986).

Dunes, water and bare land.- The default values included in N-SPECT were used (NOAA 2008, USDA 1986).

5. Floodplains

Valley bottoms function as hydrological buffers that significantly affect runoff behaviour. Distinguishing valley bottoms from hillslope is an important first step in identifying and characterising sediment deposits for hydrologic and geomorphic purposes. We used CLASS software (CSIRO-CRC

2005) and our 240m DEM to identify valley bottoms (Teng et al. 2005). The algorithm operates at a range of scales and combines the results at different scales into a single multi-resolution index (MrVBf). Based on Wilkinson et al. (2009) we selected those areas with MrVBf ≥2.0 as potential floodplains. Further processing of the floodplain grid involved delimiting the presence of potential floodplains to the areas along rivers by using 2.5km (1st and 2nd order streams) and 5km (3rd order and higher) buffers around the stream network (as modelled within Sednet) to clip it.

6. Pollutant event mean concentrations (EMC) used in catchment models

Studies varied significantly in the methods used (although many didn't report methods at all), the constituents reported (most studies included N, but both TN and TP were commonly reported constituents, and only a few were comprehensive in reporting different fractions, including Din, DIP, DON and DOP). When not directly reported, we calculated TN or TP, as well as DIN and DIP, based on reported fractions. In many cases it was not possible to know if reported values were true EMC or other types of concentrations (see Bartley et al. 2011 for definitions and methods for estimating EMC).

Following, we present a summary of the values used for modelling in N-SPECT and SedNet (**Table 5**), as well as the rational underlying our selection/calculation of values. We incorporated the concentrations used to construct management/best practices scenarios. These values were set to represent maximum potential reductions in nutrient runoff based on reported reductions from literature. For cropland classes, the value represents a reduction of 40% in EMC concentrations following fertilizer best practices. Since potential reductions are higher for areas were fertilizer use is more intense, we gradually reduced concentrations to match the cropland class (very high: 40%; high: 30%; moderate: 20%; Low: 10%; Very low: 5% reduction). We assumed that agricultural practices will have minor effects on DON and DOP concentration and thus EMC values for these fractions were not modified. For urban areas it corresponds to a maximum 20% reduction. For pasture, the considered management practices possibly have minor impact on nutrient concentrations, but can help to reduce erosion; same as for agriculture (these modifications were incorporated by modifying the [3]cover factor "C' used in RUSLE).

<u>Rationale for selection/calculation of EMC values</u>

Urban: We selected high density residential/dense urban, because most areas classified as urban areas in the land-cover/vegetation maps correspond to these classes and industrial/commercial, not to rural or sparse urban areas. When data was not available for the nutrient fraction/constituent we used

[3] Cover factor was also used to modify the weighting surface when projecting potential erosion from new urban areas under development. Since erosion from existing urban areas is relatively small, the cover factor assigned for past and current scenarios is relatively small; however, erosion in areas under development can be significant, so the highest C factor (i.e., bare land = 0.700) was assigned to these areas. For best management practices, we assumed a maximum reduction of 30% from cropland and 20% from pasture, thus used 0.1827 and 0.184, respectively.

aggregated values as noted. Since this class is the one with higher EMC values, we consider this to be an adequate/conservative approach. However, a nation-wide study in the USA shows that differences between classes are minimal and not significant. Also, since many urban areas have "deficient" sewage systems, we would expect that using a class with relatively higher EMC values would partially compensate for the lack of point source data.

Cropland (5 classes): There are significant differences in EMC values from different crops, mostly as a result from differences in fertilizer use. Since we don't have a map that identifies the crop types within the cropland classes for each period (1973, 1993 and 2000), we overlaid a map that classifies municipalities based on the average use of agrochemicals (fertilizers and pesticides) in five levels, from very low to very high (SEMARNAT-INE 2000) with the cropland areas from each land cover map. We used this classification as a proxy for fertilizer use and to subdivide the class corresponding to cropland in five subclasses. Following, we calculated the 50th, 60th, 70th, 80th, and 90th percentiles of all the available EMC values to represent the classes with different levels of fertilizer/agrochemicals use. When calculated values were the same for different cropland classes or when values were lower than those for natural vegetation classes, these were adjusted accordingly.

Grassland (natural grasslands): We used the median value of vegetation identified as natural grasslands of the full dataset, except in cases when data was not available for a particular constituent or when values between natural and modified/human-induced grasslands were minimal or inverted (i.e., higher for natural conditions). For both the Australian and North American datasets, this class sometimes also contained cattle grazed areas. For the USA it includes grassland/herbaceous, and for Australia it includes savannah woodland/ dryland sheep/cattle grazing in natural conditions, all of which have a dominant grassland understory.

Pasture (modified/induced grasslands): We used the median value of vegetation identified as modified grasslands of the full dataset, except in cases when data was not available for a particular constituent or when values between natural and modified/human-induced grasslands were minimal or inverted (i.e., higher for natural conditions). Classes in the USA include pasture/hay, and for Australia, it also includes dairy cow pasture grazing and intensive grazing areas.

Forests: Due to difficulty in matching/cross-walking our land cover/vegetation classes to diverse forest classes reported in EMC studies and to high similarity and inconsistencies in the variation in EMC values between different types of forests for different constituents we decided to aggregate EMC values for all types of forests, including dry scrub/shrub and xeric vegetation. We thus assign gradually lower values, starting from temperate forests, down to xeric/desert environments, based on calculated percentiles (i.e., 90th to 25th). Higher values correspond to woodlands (temperate coniferous and mixed forest), followed by rainforests (humid tropical forests), dry forests (thorn forests and

bushland), and scrub/shrub (xeric desert and semi-desert vegetation). We assigned the values of “rain forest” to “palm forest” due to its proximity and due to the minimal area that it occupies within the study region. Until we have more information specific for Mexico, this could be considered a working hypothesis to be tested using local experimental/observational data.

Although we would expect that erosion in areas with poor vegetation cover (e.g, scrub-shrub, xeric vegetation), would be important and therefore particulate nutrient concentrations would be higher, studies indicate that arid and xeric environments tend to export significantly less nutrients than temperate and rain forests (Perakis & Hedin 2002; Smith et al. 2003). Smith et al. (2003) estimated natural background concentrations for vegetation types that correspond fairly well with vegetation classes in our study region and showed that those corresponding to arid environments are significantly lower than those corresponding to temperate forests.

Other land cover/vegetation classes: Information for EMC values for other land cover/vegetation classes (e.g., wetlands, coastal dunes) was very scarce and values very contrasting. Considering they represent a minor portion of the region and that expected nutrient runoff from these areas is very low, either because runoff is very low or because they act like reservoirs for nutrients from upstream sources, we set values to zero for these classes. In contrast, values for bare/exposed land (i.e., cleared areas, mostly for urban or agricultural areas) were set high based on available studies, mostly only form North American studies.

Table 5. Nutrient runoff concentrations (mg/L) used for catchment models in the Gulf of California region

Grey numbers are the maximum and minimum values contained in the reviewed studies.

Red numbers indicate the criteria to assign values were adjusted, e.g., when percentile values were the same for different classes.

Class	TN	DIN	DON	TP	DIP/FRP	DOP
Urban	[4]All (median high density residential) **2.020 / 1.616** (1.200-2.220)	All (median high density residential) **0.794 / 0.635** (0.200-1.650)	[5]AUS (median aggregated) **0.300 / 0.240** (0.270-0.300)	All (median high density residential) **0.470 / 0.376** (0.200-0.570)	All (median high density residential) **0.312 / 0.2496** (0.143-0.480)	AUS (median aggregated) **0.014 / 0.011** (0.010-0.020)
Cropland (5 classes)	All (50th to 90th percentiles) Very high…**6.438/3.863** High……..**5.409/3.786** Moderate....**3.806/3.045** Low……...**2.680/2.412** Very low….**2.340/2.223** (1.100-41.497)	All (50th to 90th percentiles) Very high...**1.500/0.900** High……**1.047/0.733** Moderate...**0.850/0.680** Low……**0.750/0.675** Very low...**0.700/0.665** (0.369-2.000)	AUS (50th to 90th percentiles) Very high...**0.300** High……**0.275** Moderate...**0.250** Low……...**0.200** Very low….**0.200** (0.200-0.502)	All (50th to 90th percentiles) Very high....**1.348/0.809** High……**1.134/0.794** Moderate…**0.420/0.336** Low……**0.371/0.334** Very low….**0.340/0.323** (0.140-2.294)	All (*ad hoc* values) Very high....**0.090/0.060** High……**0.080/0.056** Moderate…**0.070/0.055** Low……**0.060/0.054** Very low….**0.050/0.048** (0.010-0.100)	All (50th to 90th percentiles) Very high....**0.025** High……...**0.020** Moderate…**0.020** Low……**0.019** Very low….**0.014** (0.002-0.027)
Pasture	All (modified) **2.235** (1.100-2.480)	All (modified) **0.399** (0.188-0.610)	AUS – all classes (90th) **0.327** (0.118-0.360)	All (modified) **0.355** (0.120-0.480)	All (both) **0.033** (0.013-0.050)	AUS - 1 value (modified) **0.017**
Grassland	All (natural) **1.086** (0.120-2.100)	All (natural) **0.190** (0.160-0.200)	AUS – all classes (50th) **0.225** (0.118-0.360)	All (natural) **0.277** (0.010-0.350)	All ([6]both) **0.033** (0.013-0.050)	AUS (natural) **0.011** (0.010-0.012)
Woodland	All (all classes including scrub/shrub: 90th percentile) **0.700** (0.050-0.830)	All (all classes including scrub/shrub: 90th percentile) **0.170** (0.040-0.180)	AUS (all classes including scrub/shrub: 90th percentile) **0.227** (0.072-0.257)	All (all classes including scrub/shrub: 90th percentile) **0.052** (0.006-0.070)	All (all classes including scrub/shrub: 90th percentile) **0.037** (0.004-0.050)	AUS (all classes including scrub/shrub: 90th percentile) **0.013** (0.005-0.016)
Rainforest	All (all classes including scrub/shrub: 75th percentile) **0.669** (0.050-0.830)	All (all classes including scrub/shrub: 75th percentile) **0.134** (0.040-0.180)	AUS (all classes including scrub/shrub: 75th percentile) **0.208** (0.072-0.257)	All (all classes including scrub/shrub: 75th percentile) **0.043** (0.006-0.070)	All (all classes including scrub/shrub: 75th percentile) **0.028** (0.004-0.050)	AUS (all classes including scrub/shrub: 75th percentile) **0.010** (0.005-0.016)
Dry forest/ Palm forest	All (all classes including scrub/shrub: 50th percentile) **0.501** (0.050-0.830)	All (all classes including scrub/shrub: 50th percentile) **0.100** (0.040-0.180)	AUS (all classes including scrub/shrub: 90th percentile) **0.150** (0.072-0.257)	All (all classes including scrub/shrub: 50th percentile) **0.020** (0.006-0.070)	All (all classes including scrub/shrub: 50th percentile) **0.010** (0.004-0.050)	AUS (all classes including scrub/shrub: 25th percentile) **0.009** (0.005-0.016)
Scrub-Shrub/ Halophytic	All (all classes including scrub/shrub: 25th percentile) **0.222** (0.050-0.830)	All (all classes including scrub/shrub: 90th percentile) **0.051** (0.040-0.180)	AUS (all classes including scrub/shrub: 90th percentile) **0.110** (0.072-0.257)	All (all classes including scrub/shrub: 25th percentile) **0.011** (0.006-0.070)	All (all classes including scrub/shrub: 75th percentile) **0.007** (0.004-0.050)	AUS (all classes including scrub/shrub: 10th percentile) **0.007** (0.005-0.016)
Wetland / Water	0	0	0	0	0	0
Dunes	0	0	0	0	0	0
Bare land	NA **1.500** (0.970-7.550)	NA **0.665** (0.540-0.790)	0	NA **0.200** (0.100-0.590)	NA **0.030** (Only value)	0

Reviewed studies for EMC values

Adamus & Bergman (1995); Mean runoff concentrations; Reference: Adamus, C. L. and M. J. Bergman (1995). "Estimating Nonpoint Source Pollution Loads with a GIS Screening Model." Journal of the American Water Resources Association 31(4): 647-655.

Baird & Jennings (1996); EMC median values; Reference: Baird, C. and M. Jennings (1996). Characterization of nonpoint sources and loadings to Corpus Christi Bay National Estuary Program Study Area. Austin, Texas, Texas Natural Resource Conservation Commission.

Bartley & Spears (2010; 2011); EMC median values (>90% indicated land use); Reference: Bartley R, Speirs W, Ellis T, Waters DK. 2011. A review of sediment and nutrient concentration data from Australia for use in catchment water quality models. Marine Pollution Bulletin .

Bartley & Spears (2010; 2011); EMC median values; Reference: Bartley R, Speirs W, Ellis T, Waters DK. 2011. A review of sediment and nutrient concentration data from Australia for use in catchment water quality models. Marine Pollution Bulletin .

[4] All = Values used were taken from full dataset, which included mostly studies from North America and Australia.

[5] AUS = When values were not available from North American studies or were not reliable or corrected (e.g., for forested areas where atmospheric deposition is expected to be high and therefore values were much higher than those reported for pristine systems), values from Australia and Americas (not USA) were used.

[6] Values for natural grassland and pasture were very similar (higher for grassland), so we used the same value for both.

Brodie & Mitchell (2005): We calculated average from ranges): Reference: Brodie J, Mitchell A. 2005. Nutrients in Australian tropical rivers: changes with agricultural development and implications for receiving environments. Marine and Freshwater Research 56: 279-302

Brodie et al. (2003); EMC 80th percentile values; Reference: Brodie J, McKergow LA, Prosser IP, Furnas M, Hughes AR, Hunter H. 2003. Sources of Sediment and Nutrient Exports to the Great Barrier Reef World Heritage Area ACTFR Report No. 03/11, Australian Centre for Tropical Freshwater Research, James Cook University - CSIRO Land and Water, Townsville, QLD.

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EPA (1983); EMC 90th percentile values; Reference: U.S. Environmental Protection Agency (EPA). 1983. Results of the Nationwide Urban Runoff Program, Volume I - Final Report. Water Planning Division, Washington, D.C.

EPA (1983); EMC median values; Reference: U.S. Environmental Protection Agency (EPA). 1983. Results of the Nationwide Urban Runoff Program, Volume I - Final Report. Water Planning Division, Washington, D.C.

Furnas (2003): EMC median concentrations; Reference: Brodie J, Mitchell A. 2005. Nutrients in Australian tropical rivers: changes with agricultural development and implications for receiving environments. Marine and Freshwater Research 56: 279-302.

Hollinger & Cornish (2002); EMC median values; Reference: Drewry JJ, Newham LTH, Greene RSB, Jakeman AJ, Croke BFW. 2006. A review of nitrogen and phosphorus export to waterways: context for catchment modelling. Marine and Freshwater Research 57: 757-74.

Lewis et al. (2002): References: Lewis WM, Melack JM, McDowell WH, McClain M, Richey JE. 1999. Nitrogen yields from undisturbed watersheds in the Americas. Biogeochemistry 46: 149-62; and Lewis WM. 2002. Yield of nitrogen from minimally disturbed watersheds of the United States. Biogeochemistry 57: 375-85.

Line et al. (2002); EMC mean values; Reference: Line DE, White NM, Osmond DL, Jennings GD, Mojonnier CB. 2002. Pollutant Export from Various Land Uses in the Upper Neuse River Basin. Water Environment Research 74: 100-08.

Line et al. (2002); EMC median values; Reference: Line DE, White NM, Osmond DL, Jennings GD, Mojonnier CB. 2002. Pollutant Export from Various Land Uses in the Upper Neuse River Basin. Water Environment Research 74: 100-08.

L-THIA (default values): EMC values; Reference: Nejadhashemi AP, Mankin KR. 2007. Comparison of Four Water Quality Models (STEPL, PLOAD, L-THIA, and AVSWAT-X) in Simulating Sediment and Nutrient Dynamics in a Watershed. Presented at ASABE Annual International Meeting, Minneapolis, Minnesota

Meixler & Bain (2010): EMC values taken from other studies and adjusted; Reference: Meixler M, Bain M. 2010. A Water Quality Model for Regional Stream Assessment and Conservation Strategy Development. Environmental Management 45: 868-80.

Moller et al. (2005): concentrations; Reference: Moller A, Kaiser K, Guggenberger G. 2005. Dissolved organic carbon and nitrogen in precipitation, throughfall, soil solution, and stream water of the tropical highlands in northern Thailand. Journal of Plant Nutrition and Soil Science-Zeitschrift Fur Pflanzenernahrung Und Bodenkunde 168: 649-59.

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7. Rainfall and precipitation settings

Precipitation

In N-SPECT rainfall is used directly to calculate runoff depth based on the Curve Number method (Eslinger et al. 2005). SedNet uses rainfall when regionalising runoff at subcatchment level (Wilkinson et al. 2008). While it is possible to estimate event-based or monthly runoff in N-SPECT it is preferable to work with annual mean (NOAA 2008). It is also advisable to use a layer built from monthly and annual rainfall records for stations which has been interpolated as grids based on precipitation records and the WorldClim dataset fits this criterion. In order to use this grid in N-SPECT we resampled the grid to 240m and converted units into inches [P(in) = P(mm)*(0.03937)].

Rain days

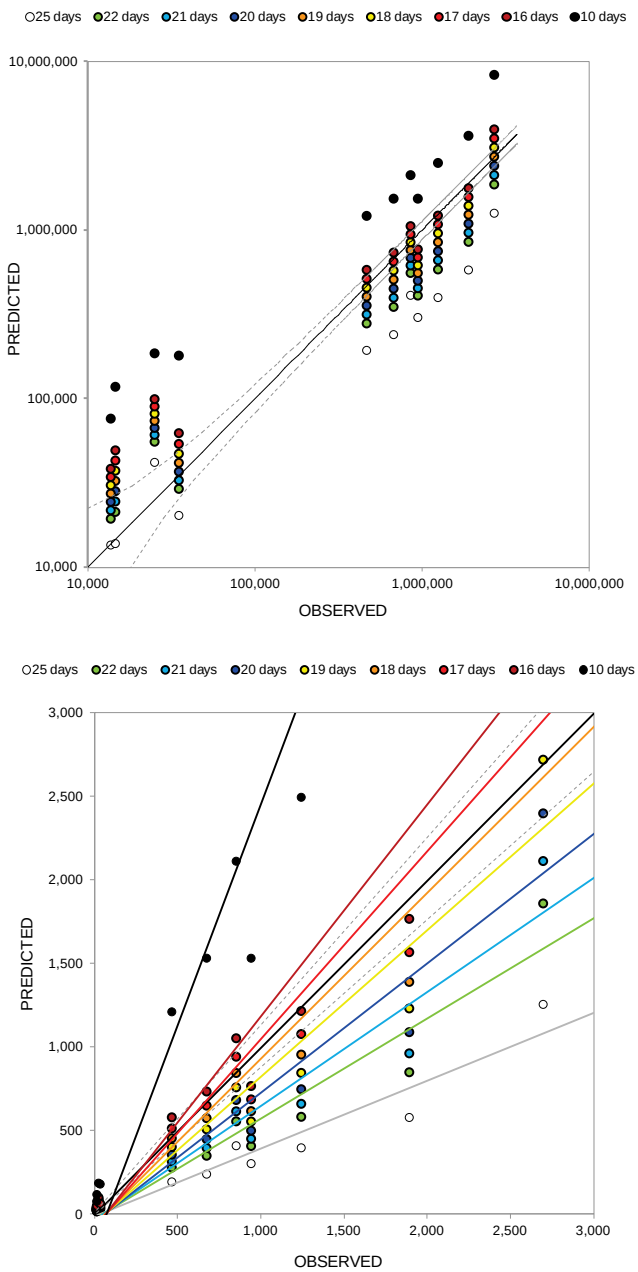
This parameter represents the average number of days/year that have rain events that produce runoff and is used to adjust the initial abstraction – Ia[7] (i.e., water retained in depressions, taken up by vegetation, evaporated and infiltrated) to account for multiple-event rainfall over the course of a year (Eslinger et al. 2005). This parameter is not entered into the model as a spatially-variable grid and is set as a single value for each run. Due to the size of the study region and variation in rainfall/runoff associated with the marked climatic variability within the region (**Figure 1**), we divided the study region in three sections and calculated this value for each of three sections of the model domain, defined based on the CN equation (USDA 1986). Modelling individual sections provided significantly better results, but increased processing time and required multiple trials to adjust values for each section to ensure that modelled runoff values were within observed values for gauges within each section and for the whole catchment when integrating upstream and/or downstream sections.

We calculated the mean (35 days) and standard deviation (16 days) of runoff days for all the basins together, and calculated runoff results based on Mean and Mean - 1STD. Based on the results of this first comparison, we calibrated the model by adding and subtracting days to get the best fit possible.

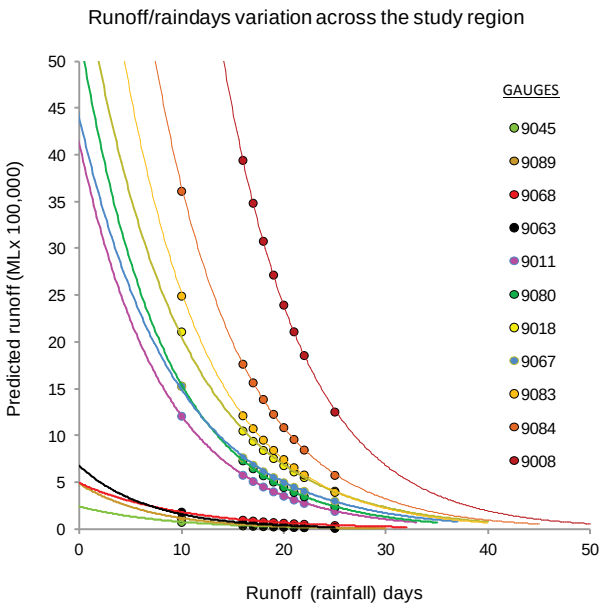
[7] Ia consists of interception, initial infiltration, surface depression storage, evapotranspiration, and other factors, was generalized as 0.2S based on data from agricultural watersheds (S is the potential maximum retention after runoff begins).

The Mean - 1STD (19 days) actually gave a very good fit, but we slightly adjusted to improve fit (18 days). Then we calculated this value for sections, accounting for natural variability in runoff days, based on variance of this parameter across subcatchments, and grouping these trying to minimise variance and the number of sections to be modelled separately. Based on the estimated rainfall days (all basins), we calculated changes in runoff by adding/subtracting days, which in turn we used as a reference to calibrate[8] the model (**Figure 1**).

Figure 1. Comparative analyses for calibration of the runoff rain days parameter



[8] It isn't recommended that users simply adjust this value to calibrate N-SPECT to local conditions, but users are encouraged to employ creative methods to accurately reflect spatial variation in raining days.



Following are the steps to calculate the number of rain days that produce runoff, based on the CN equations:

Create curve number layer: Use ArcCN-Runoff extension (Zhan & Huang 2004) to create a curve number surface based on land cover (1 for each year), soil type (HSG from INEGI) and the cross-walked CN/HSG table developed based on the USDA reference values. Convert CN field to FLOAT, and convert to Raster (240m, snapped to DEM240Agree).

Calculate Runoff Depth – **Q** (mm): Based on CNA's annual mean runoff (runoff values for a few coastal catchments were not available) – and reported catchment area – calculate runoff in a new field (FLOAT) depth based on the following equation: **Q = runoff/catchment area**; first, units were transformed into m^3. The value can be directly compared with CONABIO's "Annual Q" isolines map, which ix expressed in mm. The map broadly matched the isolines map. Convert to Raster.

Interpolate Runoff Depth surface: In order to fill in the catchment without data, and to "smooth" the borders between different catchment for further calculations an interpolated surface was created. First, we created random points within each catchment, proportional to catchment area. Then, we used the random points to extract values from Runoff Depth raster. Finally, with the points, we interpolated a continuous surface using Krigging interpolation (default values).\

Complete CNA's catchment layer: Based on the planning domain coastline (only land), create a complete layer of catchments, using CNA's catchment boundaries. This layer will be then sued to summarize data to calculate runoff (based on interpolated layer), abstraction and retention average values for each catchment.

Create average values for each catchment: Using Zonal Statistics (as table) to calculate the average values for:

- Runoff (based on interpolated surface) – Q (mm)
- Annual precipitation (original Worldclim dataset) – P (mm)

Calculate retention and abstraction: Based on CN raster, calculate the following:

- Soil retention (mm): S=((1000/CN)-10)*24.5
- Initial abstraction (mm): Ia=(0.2*S)

Average catchment values were calculated following two different approaches and compared:

- First, the interpolated runoff raster, original annual precipitation, and calculated retention and abstraction were used to calculate runoff rain days for each cell using the formula shown below; Zonal Stats based on catchment was then used to calculate average value for each catchment:
- Second: Based on original layers (not average/catchment) calculate
- Following, join calculates tables based on catchment ID to "Complete CNN catchment" layer, and export as new Shapefile with the new statistics values for each variable.

Raindays_cell = ([Runoff_krg] / [abstraction76]) * (1 + ([retention76] / ([ppt_goc] * [abstraction76])))

Calculate annual runoff days: Finally, based on average runoff and rainfall values, calculate the number of rain days that produce runoff using the following equations:

- Runoff days (annual): $^{*}R = (Q/Ia) \times (1 + (S/(P \times Ia)))$
- Q = Annual runoff depth (we used the maximum runoff depth based on the annual natural runoff at for each catchment from CNA database (2008), validated with a coarse-scale national isolines map)
- P = Annual precipitation (mm)
- S = ((1000/CN) – 10)*24.5; soil retention (mm)

The equation for runoff days was derived from USDA curve number equation for annual runoff depth:

8. Rainfall erosivity

The erosivity (R factor) has been shown to be the index most highly correlated to soil loss at many sites throughout the world. It refers to the sum of individual storm E1-values for a year averaged over long time periods (> 20 years) to accommodate apparent cyclical rainfall patterns. The E1 term is an abbreviation for energy multiplied by the maximum intensity in 30min (Renard & Freimund 1994).

In absence of detailed and long-term information an approach to estimate erosivity at regional and national scales is to regionalize the parameter based on precipitation distribution and occurrence characteristics (duration, intensity, number of events) according to historical record available from climatological stations (Diodato & Bellocchi 2007; Mikhailova et al. 1997; Millward & Mersey 1999). Estimation relations established for similar climatic zones could be considerably in error, and the predicted soil loss may be far from exact, but they may be the best available for at least assessing the erosion potential or relative erosion rates from different conditions or soils (Renard & Freimund 1994). In Mexico, Cortés-Torres (1991) typified this parameter for 14 regions (based on data from 53 climatological stations and 4-11 years) and developed a regression equation for each region, based on annual precipitation based on the following general second order regression equation:

$$R = A \cdot P + B \cdot P^2$$

Where: R is the erosivity factor [MJ mm/(ha h)/year]; A and B are the estimated coefficients for the regression equations and P is annual precipitation [mm/year]. Erosivity units can be converted into the US system (as required by N-SPECT) by dividing R by 17.2.

SEMARNAT-UACH (2002) made a validation and revision of the equations proposed by Cortés-Torres (1991) and updated by Figueroa et al. (1991) based on updated data and introducing topography, and propose to use the original equations to some regions, use the equations form some regions on other that are closely related or use a general equation instead [originally proposed by Cortés-Torres (1991) as a general equation for Mexico] for those regions where major differences were found in their analyses (**Figure 2**).

We used the 1km annual precipitation grid to estimate the R-factor based on the regionalized erosivity map and regression equations proposed by Cortés-Torres (1991), as updated by Figueroa et al. (1991) and modified by SEMARNAT-UACH (2002). Five of the 14 erosivity regions described by Cortés-Torres (1991) occur within our study region; **Table 6** shows the regression equations, as well as the calculated R^2. Once each erosivity was calculated for each section according to revised formulas, layers were merged and the new layer was randomly sampled to produce a new “smoothed” erosivity surface to avoid abrupt changes between sections (sampling points were distributed proportionally within each erosivity section, based on area; i.e., #pts = Area (km^2)/100; 100 out of the total 19,550 points fell outside the raster and were eliminated). The new surface was calculated based on Kriging interpolation.

Figure 2. Erosivity regions of Mexico

Table 6. Regionalised regression equations used to calculate erosivity

Region	Equation[#]	R^2
II	$3.45552 \cdot P + 0.006470 \cdot P^2$	0.93
IV	$2.89594 \cdot P + 0.002983 \cdot P^2$	0.92
VI	$6.68471 \cdot P + 0.001680 \cdot P^2$	0.90
V, VII	$0.71508 \cdot P^{1.30751}$	0.88
X	$6.89375 \cdot P + 0.000442 \cdot P^2$	0.95

9. Evapotranspiration

We compared the application of two different evapotranspiration datasets. The first was modelled based on the Turc method, using global high-resolution (30 arc seconds) climate dataset (Hijmans et al. 2005) available online from WorldClim (http://worldclim.org/). The data required to estimate PET based on another commonly used method (e.g., Priestley-Taylor) was not available. We used the Global Potential Evapotranspiration (Global-PET) dataset (available online from CGIAR: http://www.cgiar-csi.org), calculated based on the Hargreaves evapotranspiration equation (Trabucco et al. 2008) using WorldClim datasets. This method has been recommended for regional applications of PET (Lu et al. 2005).

The equations to estimate evapotranspiration based on the above mentioned methods are:

a) Turc-method

E = P / [SQRT(0.9 + (POWER(P,2) / POWER(L,2)))]

L = 300 + (25 * T) + 0.05 * POWER(T,3)

E: Evapotranspiration (mm)

P: Annual mean precipitation (mm)

T: Mean annual temperature (°C)

b) Global-PET (Hargreaves 1994)

$PET = 0.0023 \times RA \times (T + 17.8) \times TD^{0.5}$

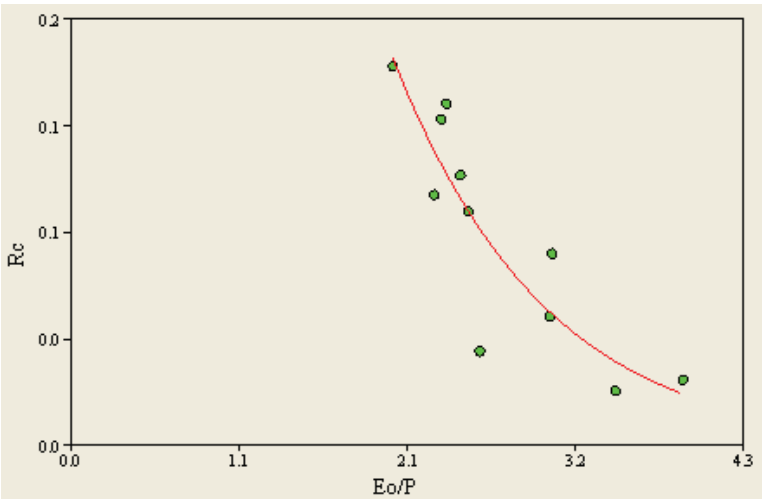
PET: Potential evapotranspiration (mm/month)

T: Mean temperature (C°)

TD: Daily temperature range (C°)

RA: Extra-terrestrial radiation on top of atmosphere as equivalent of evaporation (mm/month)

The following graphs shows a good fit between mean annual runoff coefficient (Rc) and PET/P, based on the selected PET dataset ($E^* = 0.7759$).



10. Hydrometric data processing

We used monthly and daily records from gauging stations data contained in the BANDAS database (IMTA 2010). We selected 232 stations (11 for Yaqui) based on the number of years recorded (≥20 years, 15 years for areas with few stations) and continuity of records (≥10 years of continuous records) in order to obtain reliable discharge estimates (Wilkinson et al. 2006). To calculate unregulated and regulated scenarios in SedNet we divided the dataset on two sections: unregulated and regulated (i.e., stations were flow is affected by reservoir). Records from gauge stations that preceded dam construction/operation were identified as unregulated, while records that spanned the periods before and after dam started operating were divided. We also identified stations with TSS records and summarised annual sediment loads using monthly data. Due to inaccuracies in the database, manual verification of the geographic position of each gauge station was required to validate its position in relation to the mapped river network (CONABIO 2008) and to adjust its location when needed.

Following are the steps for gauge data preparation:

- Download Mexico's gauge stations full catalogue (2,248 gauge stations).

  http://www.imta.gob.mx/index.php?option=com_wrapper&view=wrapper&Itemid=145

- Eliminate stations with no geographic data and those located in the USA (1,563 stations).
- Map stations and eliminate those located outside Mexico or in the sea (1,551 stations).
- Clip to planning domain, based on coastal catchment boundaries (323 stations).
- Identify stations that can potentially be used for regionalization analysis (232 stations).
  - Verify number of years recorded: select stations with minimum **20 years** (**15 years for areas with few stations**) and assign a record score[9]; based on this criterion we eliminated
  - Continuity of records: preferably select stations with **≥10 years of continuous records**[10]; and **>300 days accurate recording/year** (verify w/Sednet) to get a reasonable mean discharge estimate; stations that did not fulfill these criteria were eliminated[11] **(259 stations).**
  - Identify regulated and un-regulated stations[12], and classify them based on upstream reservoir(s) size.
  - Mark stations with predominantly **old records** (i.e., <mean #years – STD); for areas were large reservoirs exist, consider including the <u>unregulated portion of records</u> from gauges downstream of where reservoirs now exist.

[9] REC_SCORE: 1 =≥20; 2 = [15-20); 3 = [10-15); 4 = <10 years); deleted stations, i.e., stations w/records under 20 or 15 years are identified as set 2.

[10] 10_YEARS: ≥10 years of continuous records = 1; <10 years = 0; deleted stations are identified as set 1.

[11] Monte Carlo simulation could be used to create a distribution to fill in the blanks for stations with few records, and because daily records are needed, estimate these from monthly or annual flows. A daily rainfall/runoff model might be also appropriate, but that would require calibration. In the first instance these gauges should be excluded from the analysis.

[12] REGULATED:0 = No reservoirs; 1 = Very large; 2 = Large; 3 = Medium ; and 4 = Small reservoirs

- Daily discharge records (not monthly or annual); MAF regionalization could take a daily time-series =1/30*monthly discharge if required. However, bank erosion and more especially floodplain deposition may be less accurate using this approach.

- Sort stations in two groups (unregulated and regulated flow datasets): Identify and mark as **regulated stations** those located downstream of large (>10 has, 0.1 km2) reservoirs (however, if pre-reservoir records, these portion can be included in the unregulated folder – in this case the shapefiles of unregulated and regulated should both have the reservoir gauge point included).

  Data transformation: ID = (SD x 86400) x 0.001
  SD = Bandas Source Data (m3/s)
  ID = SedNet Input Data (Ml/day)

- Identify double/segmented records (e.g., stations measuring flow from left/right margins of the river/reservoir; relocated stations).

- Identify gauge stations that also have records for TSS (sediment sampling0.

- Identify any location problems in gauge location (based on original coordinates); If any problems are encountered based on the drainage area comparison in Sednet, explore the station‘s attributes table (i.e., measured current) and the 1:250,000 map of rivers try to identify the corresponding stream link and – if needed – move it to match the corresponding river link.

11. Reservoirs

We mapped 11 (133 for all GoC catchments) reservoirs by combining the national reservoir database (CONAGUA 2008) and the most recent landcover map (2000), in which water bodies are delineated. Information was validated and corrected (including the mapping of additional dams) using online ArcGIS map (source). 51 reservoirs with a surface of <100has were eliminated from further analysis (mapping of small reservoirs was considered to be incomplete, and their influence in terms of sediment retention is probably minimal in comparison to larger reservoirs; also volume calculations for very small reservoirs was considered to be too unreliable).

For reservoirs with no information on storage capacity we estimated volume using the following procedure: we converted the dams polygons to points (vertices). Then, we used these points to extract the altitude values from a 30m resolution aster (source). The average and STD was estimated for each dam in order to estimate a waterline (altitude) that could be used to estimate the volume. 3 different waterline heights were used: AVG, AVG + STD, AVG + 2STD, MAX. We also used the reservoirs polygons to extract the topography within these from the 30m DEM. Using the Cut/Fill tool (from ArcGIS 3D Analyst) we calculated the difference between the different waterlines (i.e., average, AVG+STD, AVG+2STD) and the topography to calculate the volume within each reservoir. Results

were compared with reservoir maximum and use capacity available from CONAGUA database. We selected AVG+2STD, because our estimated volume was within the capacity range provided by the database for most reservoirs for which we had capacity information. This estimate was only used for reservoirs for which capacity data was missing, while the 67% of NAME was used for reservoirs which available information.

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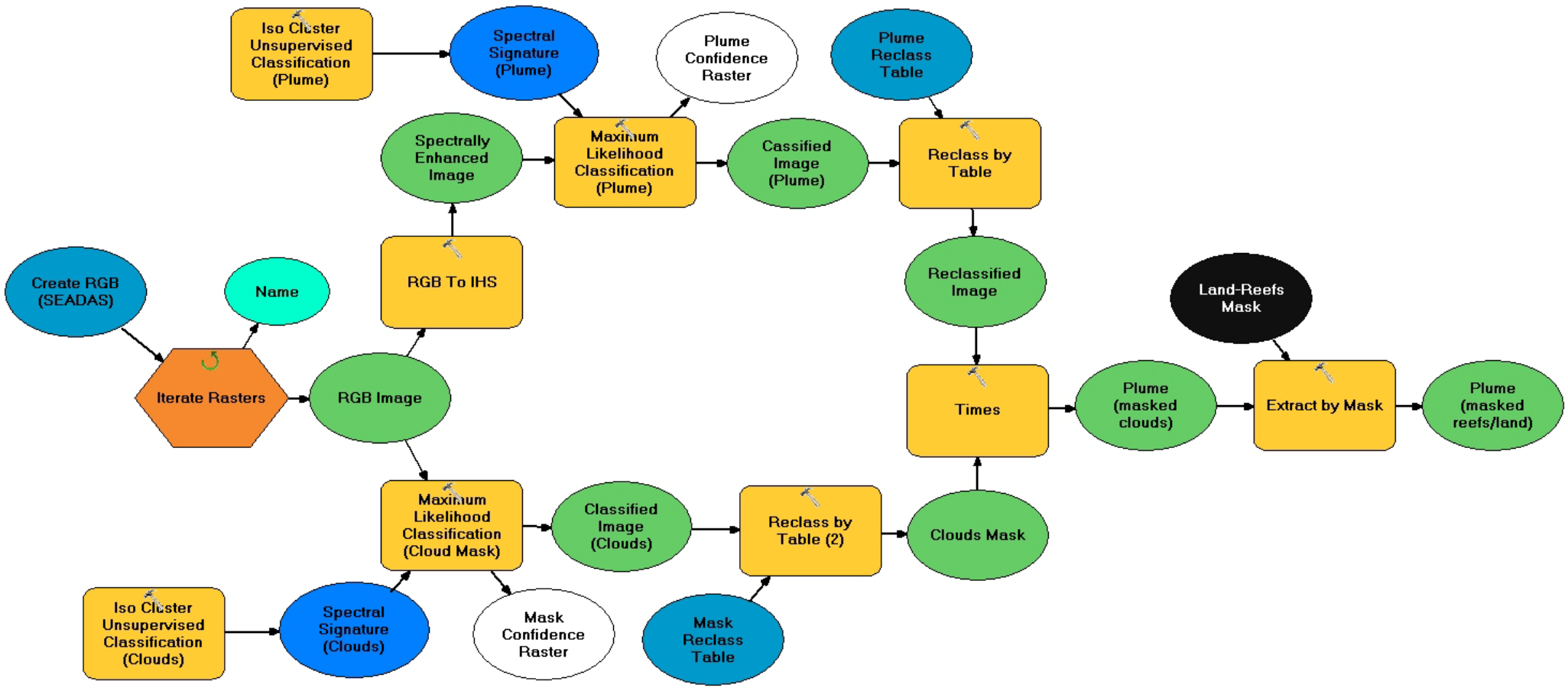
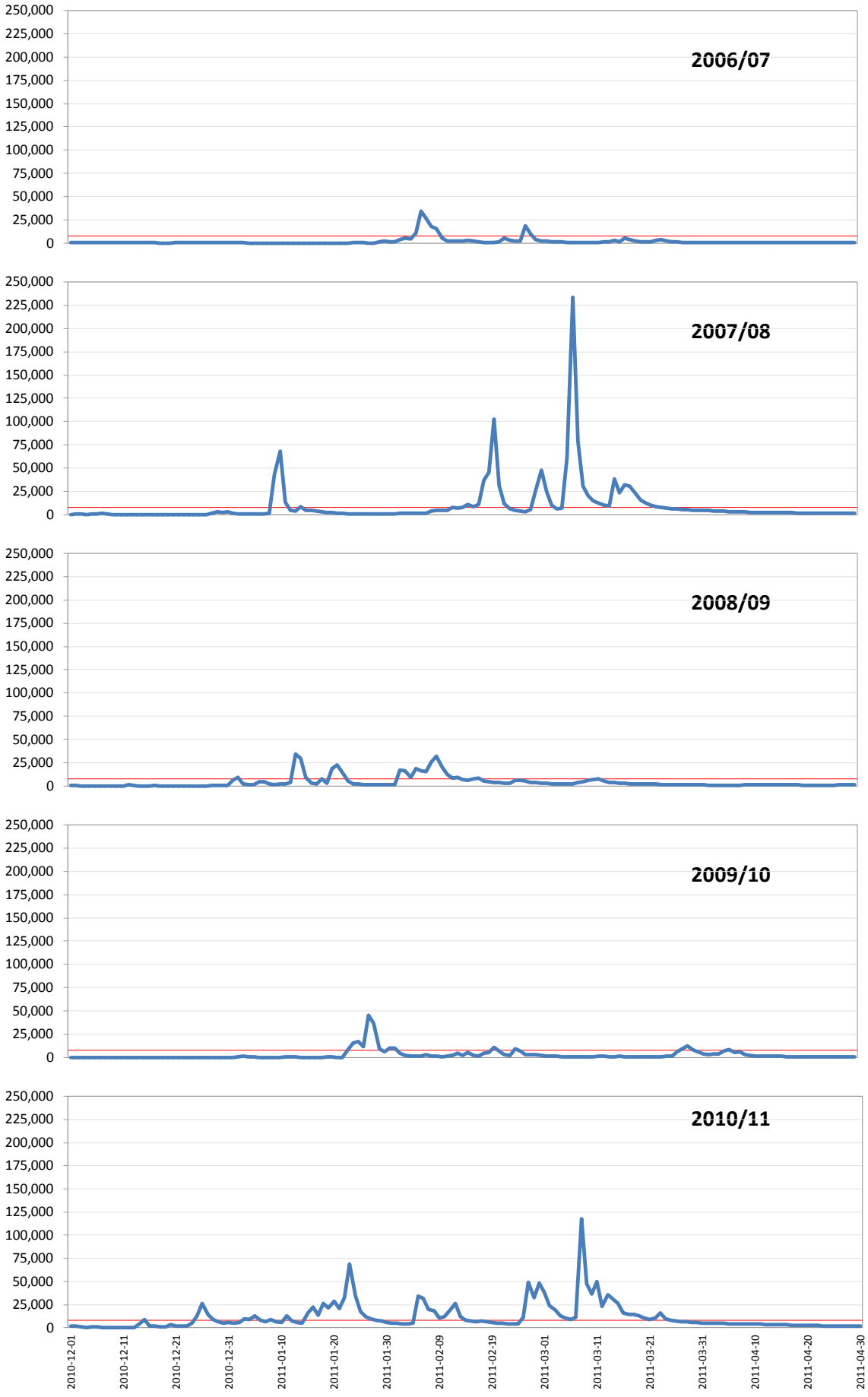
Supplementary Material C5 - Figure 2. Procedure to map plumes and cloud based on supervised classification of true-colour images



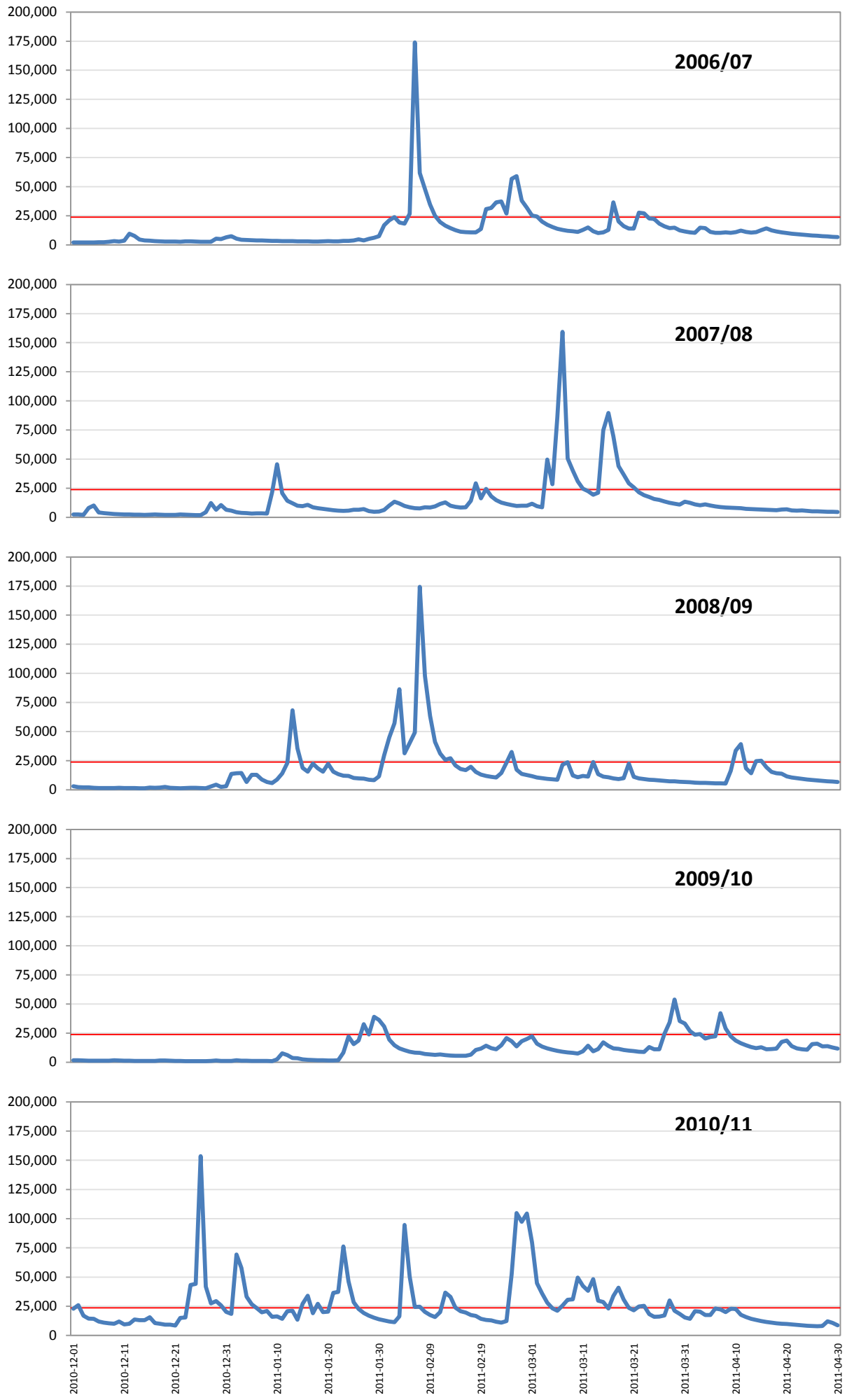
True colour images were used to identify and characterise plumes and to mask out areas considered unsuitable for classification (i.e., areas of high cloud cover and intense sun glint that resulted in errors in the assignment of plume classes). The final outputs of this stage are grids (i.e., raster formatted maps) representing: the full extent of plumes (presence/absence) and 6-class plume maps that represents a gradient in the selected L2 parameters (the gradient in L2 parameters is considered to be indicative of water quality variation within plumes and thus was used to identify the plume boundary and to represent the potential dispersal of pollutants incorporated in the qualitative assessment of plume exposure proposed in this study), as well as a map of dense-clouds and intense sun glint used to mask out areas unsuitable for colour classification.

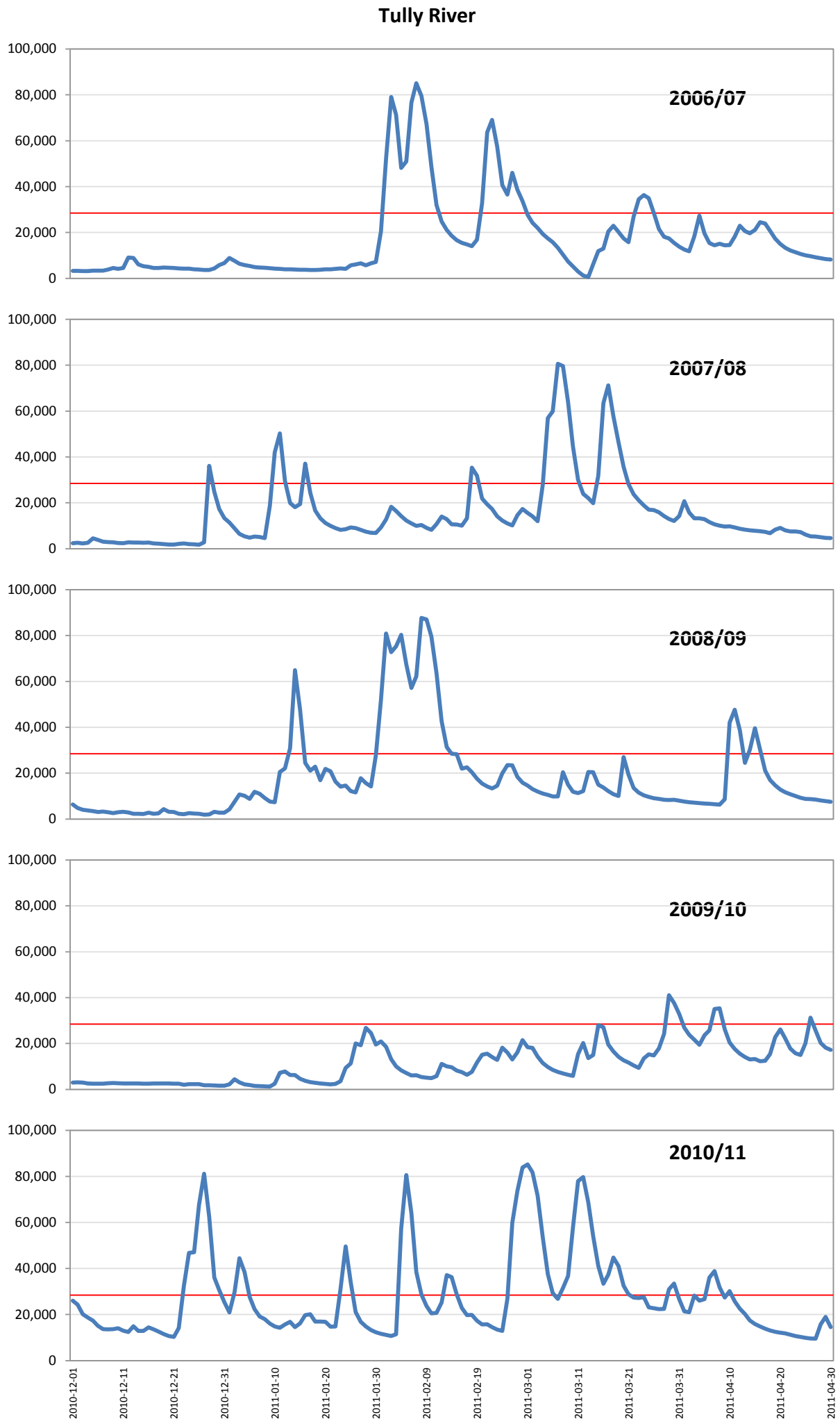
Peak events discharge: Barron River



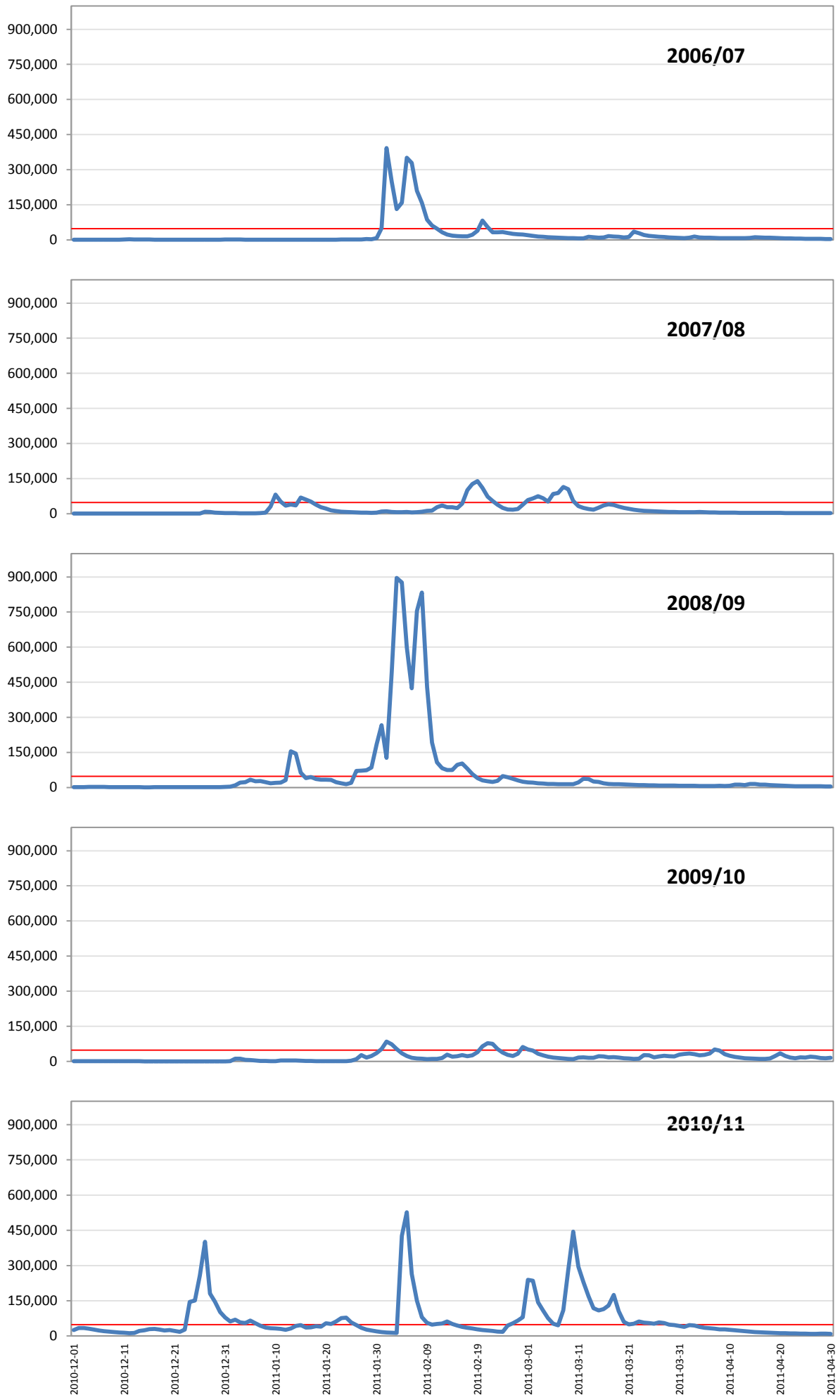
Red lines marke the 95th percentile discharge for a 10-year (2002-2011) period

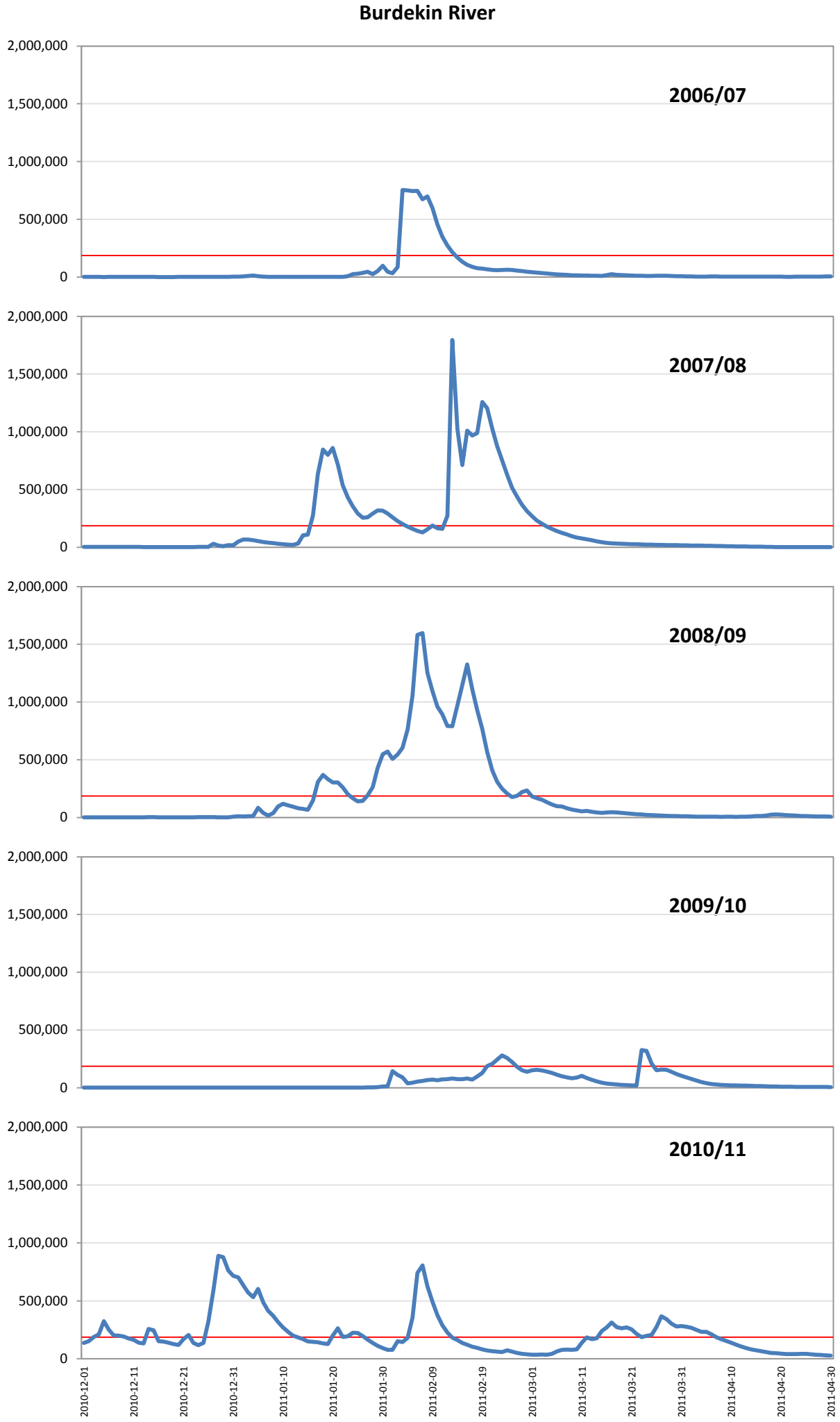
Johnstone River

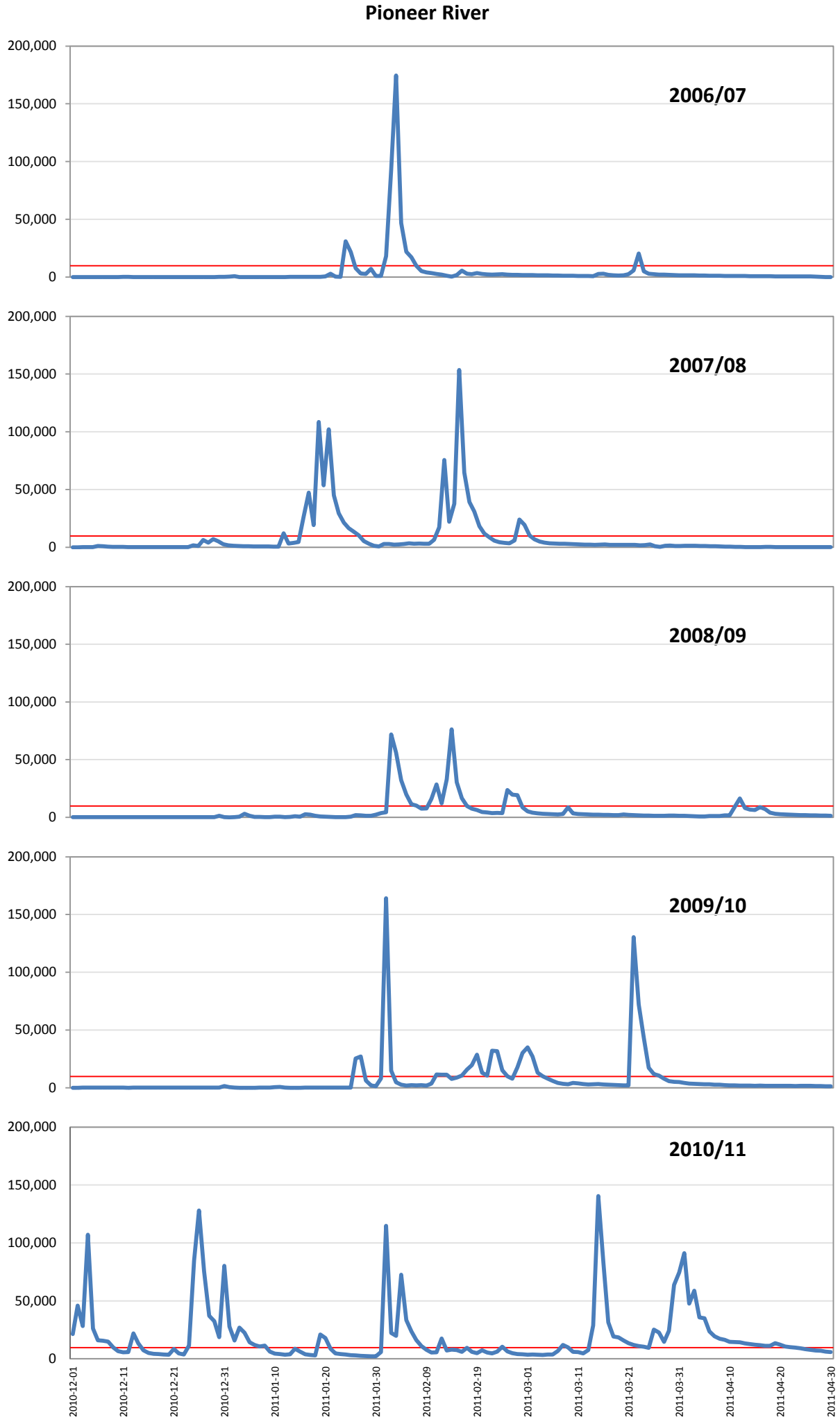



Tully River
2006/07
2007/08
2008/09
2009/10
2010/11
100,000
80,000
60,000
40,000
20,000
0
2010-12-01
2010-12-11
2010-12-21
2010-12-31
2011-01-10
2011-01-20
2011-01-30
2011-02-09
2011-02-19
2011-03-01
2011-03-11
2011-03-21
2011-03-31
2011-04-10
2011-04-20
2011-04-30

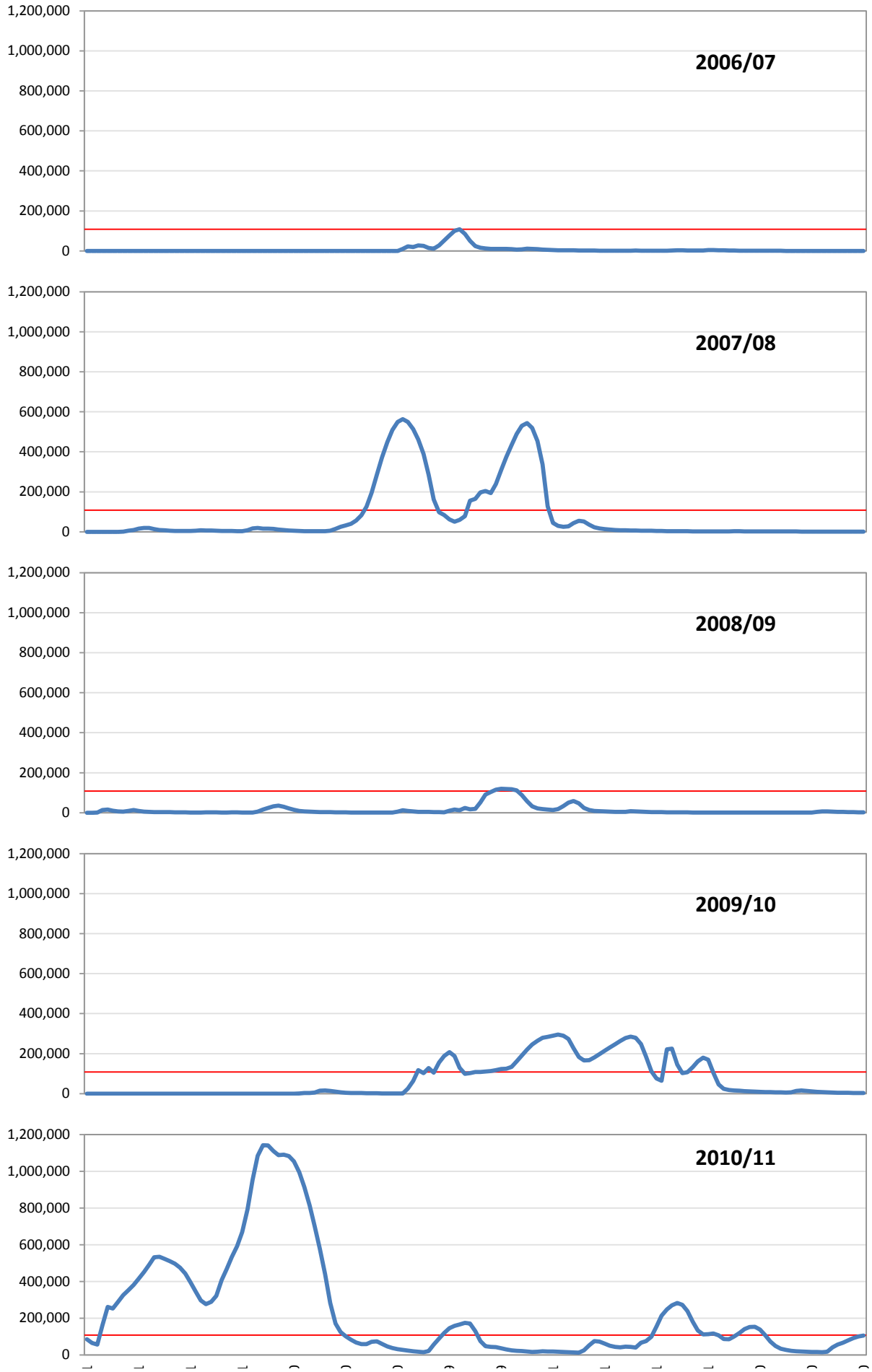
Herbert River




Burdekin River
2006/07
2007/08
2008/09
2009/10
2010/11
2,000,000
1,500,000
1,000,000
500,000
0
2010-12-01
2010-12-11
2010-12-21
2010-12-31
2011-01-10
2011-01-20
2011-01-30
2011-02-09
2011-02-19
2011-03-01
2011-03-11
2011-03-21
2011-03-31
2011-04-10
2011-04-20
2011-04-30


Pioneer River
2006/07
2007/08
2008/09
2009/10
2010/11
200,000
150,000
100,000
50,000
0
2010-12-01
2010-12-11
2010-12-21
2010-12-31
2011-01-10
2011-01-20
2011-01-30
2011-02-09
2011-02-19
2011-03-01
2011-03-11
2011-03-21
2011-03-31
2011-04-10
2011-04-20
2011-04-30

Fitzroy River



2010-12-01

2010-12-11

2010-12-21

2010-12-31

2011-01-10

2011-01-20

2011-01-30

2011-02-09

2011-02-19

2011-03-01

2011-03-11

2011-03-21

2011-03-31

2011-04-10

2011-04-20

2011-04-30