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Biodiversity offsets for moving conservation targets

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Declaration of Originality

This dissertation results entirely from my own work, and includes nothing that is the outcome of work done by or in collaboration with others except where this has been specifically indicated in the text.

Joseph Bull, March 2014

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Abstract

Conservation is difficult for moving targets, such as migratory species or landscapes subject to environmental change. Biodiversity offsetting is a novel approach that involves active compensation for biodiversity lost through development, with an objective of no net loss of biodiversity overall. In this thesis, I explore the use of biodiversity offsets for moving targets. My case study is the conservation of the migratory saiga antelope *Saiga tatarica* alongside industry in the Ustyurt plateau, Uzbekistan.

Key challenges for offsetting include: specification of an appropriate frame of reference for evaluating no net loss, determination of requisite ecological gains, and the degree of flexibility permitted in biodiversity trades.

I use bespoke simulation models to predict whether no net loss of biodiversity can be achieved within various hypothetical frames of reference, i.e. against different socio-ecological baselines and counterfactual scenarios. The reference frame determines the feasibility and effort required in achieving conservation objectives, and I shed light upon those ecosystem dynamics for which offsets may be appropriate. I develop a socio-ecological counterfactual for saigas and their Ustyurt habitat, relying upon satellite imagery and secondary data sets. Even with limited data, it proves possible to develop an instructive counterfactual for intervention.

To calculate offset requirements, I first quantify impacts of industrial activity on the Ustyurt. Vegetation impacts are measured, mapped and projected to the landscape scale, and the influence on mobile species such as saigas is considered. Via quantitative comparison, I show that the application of different available offset calculation methodologies to these data – which all purport to achieve no net loss of biodiversity – would result in divergent offset requirements. This implies that offset methodologies should be tailored to specific moving target problems, rather than generalised.

Finally, I use conservation planning software to compare the performance of flexible and non-flexible offsets. Zonation is used to model the effect of permitting flexibility in the biological, spatial and temporal constraints placed upon offsetting, and RobOff to assess the optimum return on investment under uncertainty. I find that a mixture of flexible and non-flexible offsets is desirable for conserving moving targets in the Ustyurt.

We must give deeper consideration to the dynamic nature of ecosystems when designing conservation interventions. Biodiversity offsets have potential in this regard. To realise the potential, we should specify appropriate frames of reference, tailor metrics, and consider allowing flexible biodiversity trades.

Реферат статьи

Движущиеся цели, такие как мигрирующие виды и ландшафты, находящиеся под воздействием изменений в окружающей среде, сложно сохранять. Компенсация биоразнообразия являет собой новый подход, предусматривающий активную компенсацию потерь биоразнообразия, полученных в ходе развития, с целью исключения остаточных потерь для биоразнообразия. Данная работа рассматривает вопросы компенсации ущерба биологическому разнообразию для движущихся целей. Предметом исследования является сохранение миграционного сайгака *Saiga tatarica*, наряду с промышленным развитием плато Устюрт в Узбекистане.

Ключевые вопросы компенсации включают выбор соответствующей системы оценки с целью исключения остаточных потерь, определение потребных экологических выгод и степени допустимой гибкости в решениях по сохранению биоразнообразия.

Использованы специальные имитационные модели для определия возможности исключения остаточных потерь для биоразнообразия в рамках различных гипотетических систем оценки, т.е. для различных социально-экологических стандартов и сценариев. Система оценки определяет целесообразность и усилия, необходимые для достижения природоохранных целей, здесь также представлена динамика экосистем, в рамках которой возможна компенсация. Разработаны социально-экологические сценарии для сайгаков и их среды обитания на плато Устюрт на основе спутниковых изображений и результатов вспомогательной исследовательской базы данных. Даже при ограниченности данных, представляется возможным разработка пилотного сценария по применению природоохранных мепроприятий.

Для расчета компенсации, прежде всего осуществлена количественная оценка воздействия от производственной деятельности на Устюрте. Воздействие на растительность измерено, нанесено на карту и экстраполировано в масштабах ландшафта, а также рассмотрено влияние на мобильные виды, такие как сайгак. Посредством количественного сравнения, показанно, что применение различных доступных методик расчета этих данных, имеющих целью исключение остаточных потерь для биоразнообразия, приводит к различным требованиям к компенсации. Это указывает на то, что методологии компенсации должны быть приспособлены к конкретным движущимся целям, а не обобщены.

Наконец, использовано программное обеспечение по природоохранному планированию для сравнения эффективности гибких и негибких схем компенсации. Zonation использовано для моделирования эффекта допустимой гибкости в биологических, пространственных и временных ограничениях, накладываемых на компенсации, и RobOff использовано с целью оценить оптимальную отдачу от инвестиций в условиях неопределенности. Подчеркивается необходимость сочетания гибких и негибких коменсационных схем, для сохранения движущихся целей в Устюрте.

Предлагается принимать во внимание динамическую природу экосистем при разработке природоохранных мероприятий. В этом случае представляется целесообраным применение компенсации ущерба биологическому разнообразию. С этой целью, необходимо определить соответствующие системы оценки, индивидуальные показатели и рассмотреть вопрос о допустимой гибкости в решениях по сохранению биоразнообразия.

Acknowledgements

Не имей сто рублей,
А имей сто друзей.

Thanks first of all go to my excellent supervisors: E.J. Milner-Gulland, Blake Suttle, Paul Hotham, and Navinder Singh. E.J.: quite simply, everything I have achieved in academia so far has been because of you. Blake: your enthusiasm and intellectual curiosity are unparalleled, and have been an inspiration. Paul: your encouragement and pragmatic outlook on conservation were indispensable throughout. Navinder: your insights were often crucial and always fascinating, and I am hugely grateful for the time I spent with you and your colleagues in Umeå.

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Published papers

This thesis contains four chapters that have now been published as articles in peer-reviewed journals. The contributions made to each chapter were as follows:

Chapter 2: conceived, researched and written by JWB. KBS, AG, NJS and EJMG provided extensive review, creative suggestions and commentary.

Bull, J.W., Suttle, K.B., Gordon, A., Singh, N.J. & Milner-Gulland, E.J. (2013a) Biodiversity offsets in theory and practice. Oryx, 47(3):369-380.

Chapter 3: conceived by JWB and EJMG. Researched and written by JWB. KBS, NJS and EJMG provided extensive review and commentary.

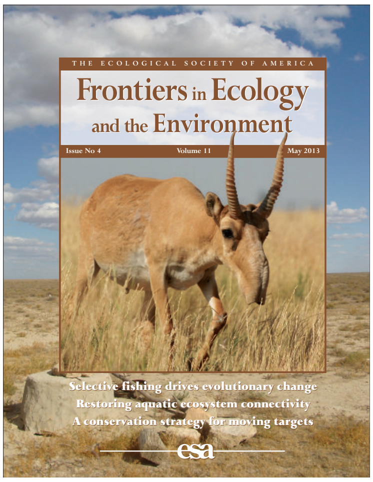
Bull, J.W., Suttle, K.B., Singh, N.J. & Milner-Gulland, E.J. (2013b) Conservation when nothing stands still: moving targets and biodiversity offsets. Frontiers in Ecology and the Environment, 11(4):203-210.

Chapter 4: conceived by JWB and AG, with support from EAL. Analytical model and overarching conceptual framework developed by JWB with support from AG and EAL. Predictive model developed by AG and adapted by JWB. Analyses carried out and results obtained by JWB, with support from AG. Written by JWB. AG, EAL, KBS and EJMG provided extensive review and commentary.

Bull, J.W., Gordon, A., Law, E.A., Suttle, K.B. & Milner-Gulland, E.J. (2014) Importance of baseline specification in evaluating conservation interventions and achieving no net loss of biodiversity. Conservation Biology. DOI: 10.1111/cobi.12243.

Chapter 6: conceived by JWB. Methodology developed by JWB and KBS, and piloted by JWB. Data collection carried out by JWB and ILJ, with support from AVE. Statistical analyses completed by ILJ. Mapping exercises completed by JWB and ILJ. Written by JWB and ILJ. EJMG and KBS provided extensive review and commentary (* denotes joint lead authorship).

Jones, I.L., Bull*, J.W., Suttle, K.B., Esipov, A.V. & Milner-Gulland, E.J. (2014) Quantifying habitat impacts of natural gas infrastructure to facilitate biodiversity offsetting. Ecology & Evolution, 4(1):79-90.*



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Chapter 1

Introduction

"For nitrates are not the land, nor phosphates; and the length of fiber in the cotton is not the land. Carbon is not a man, nor salt nor water nor calcium. He is all of these, but he is much more, much more; and the land is so much more than its analysis. The man who is more than his chemistry, walking on the earth…that man who is more than his elements knows the land that is more than its analysis."

John Steinbeck (from "The Grapes of Wrath", 1939)

1.1 Background

The conservation of global biological diversity alongside continued economic development is a key challenge for humanity in the 21st Century (UN, 2000). Human societies depend on diverse, functioning ecosystems in innumerable ways that are not fully understood (Lubchenco, 2000), yet despite the efforts of the conservation community, the variety of life on earth continues to decline (Butchart et al., 2010). Evaluating why existing conservation interventions (e.g. the creation of protected areas) have been insufficient to prevent declines, and determining how to design new or adapted interventions that are more effective, is consequently paramount.

Increasing attention has been paid to the quantitative evaluation of conservation interventions since the early 1990s (Ferraro & Pattanayak, 2006). Yet evaluation and performance reporting remain largely ineffective (McDonald-Madden et al., 2009). In particular, there is an ongoing lack of consideration given to counterfactual scenarios, i.e. what would have happened in the absence of the intervention (Maron et al., 2013). Whilst there has been extensive research interest focused upon how best to measure biodiversity (e.g. Mace & Baillie, 2007), and how to set baselines for evaluation (e.g. Willis et al., 2010; Gordon et al., 2011a), research is still needed to better establish means for measuring and evaluating conservation interventions in a rapidly changing world (Nicholson et al., 2010).

Accompanying this need to interrogate existing approaches has been a growing interest in measures beyond more conventional command-and-control type interventions – such as strictly protected areas, or bans upon the hunting and trade of certain species – and towards approaches that are based more upon incentives or market signals (e.g. Berkes, 2004; Sommerville et al., 2009; Pirard, 2012). This is not only due to the considerable difficulties in ensuring compliance to biodiversity legislation on the ground (Keane et al., 2008), but also to the relatively successful performance of market-based instruments in other environmental fields, such as pollution reduction (Godden & Vernon, 2003). However, consistently effective and theoretically sound market-based instruments for biodiversity conservation have yet to be demonstrated (Pirard, 2012).

Biodiversity conservation often has to be implemented in regions characterized by multiple economic actors with different objectives, some of which (e.g. the extractive industries) may be perceived to be incompatible with conservation activities, but which are nonetheless judged necessary by society. The result has been a wealth of literature upon, and ongoing development of decision tools for, spatial conservation planning (Margules & Pressey, 2000; Watts et al., 2009). The next challenge in this field is how to further develop conservation planning approaches in such a way as to deal with 'moving targets': i.e. regions undergoing increasingly rapid environmental change (Pressey et al., 2007), or faunal species that are highly mobile (Rayfield et al., 2008; Game et al., 2009). Equally, conservation planning is indispensible in understanding the role that the private sector has to play in biodiversity conservation – a related topic now recognized as crucial, yet upon which little has been published (Bayon & Jenkins, 2010; Houdet et al., 2012).

Biodiversity offsetting is a relatively novel conservation mechanism. Biodiversity offsetting is intended to account and compensate for biodiversity lost during development activities, is based around the creation of incentives for private sector conservation, and may be flexible enough to incorporate moving targets – thus cutting across a number of the interconnected issues raised above. Biodiversity offsets ('offsets') are an increasingly popular tool in practice, yet remain controversial amongst conservation scientists. The intention of offsetting is that, once any negative ecological impacts associated with economic development activities have been avoided and mitigated as far as possible, the residual impacts are fully compensated for (i.e. offset) through additional conservation activities, resulting in no net loss of biodiversity overall (BBOP, 2012). The popularity of offsetting lies in the potential to meet the objectives of biodiversity conservation and economic development in tandem (Gardner et al., 2013; Habib et al., 2013). The controversy lies in the need to accept biodiversity losses in return for uncertain gains (Moilanen et al., 2009; Bekessy et al., 2010; Maron et al., 2012), and in making the assumption that nature can ever be meaningfully quantified and exchanged – a point captured memorably by John Steinbeck in the quote at the start of this chapter. The offsetting approach is being widely adopted in policy and practice (Madsen et al., 2011), even whilst methodologies and the overriding conceptual framework are still in need of research and development.

A characteristic example of a moving target problem relates to the conservation of the migratory saiga antelope *Saiga tatarica* in the Ustyurt plateau, northwest Uzbekistan. Biodiversity offsets have been proposed as one of a raft of measures for maintaining areas of this semi-arid scrub-dominated landscape, a winter home to the Ustyurt saiga population, alongside increasing oil and gas activity (UNDP, 2010a). The saigas are Critically Endangered and move vast distances across the landscape (Milner-Gulland, 2010), the Ustyurt itself has undergone rapid social and environmental change largely as a result of the drying of the Aral Sea (Micklin, 2007), and there are extensive natural gas extraction activities planned in the region. Despite the proposals to use biodiversity offsets within the Ustyurt, it has yet to be established how offsets can be designed so as to be effective there, where they can best be located, and what conservation activities they should involve. It can be readily surmised, even from this brief description, that the Ustyurt case study exemplifies a range of the contemporary challenges to biodiversity conservation discussed above.

1.2 Aims and objectives

The overall aim of the research reported upon in this thesis is to improve the theoretical basis that underpins biodiversity offsetting, and clarify when and where offsets are appropriate as a tool for biodiversity conservation. In particular, I aim to explore how offsets can be implemented for moving targets and in the context of socio-ecological change. I frame this investigation using the case study of biodiversity offsets in the Ustyurt plateau, Uzbekistan, and in doing so aim to provide practical and scientifically sound policy advice to decision makers in that region.

The main objectives are:

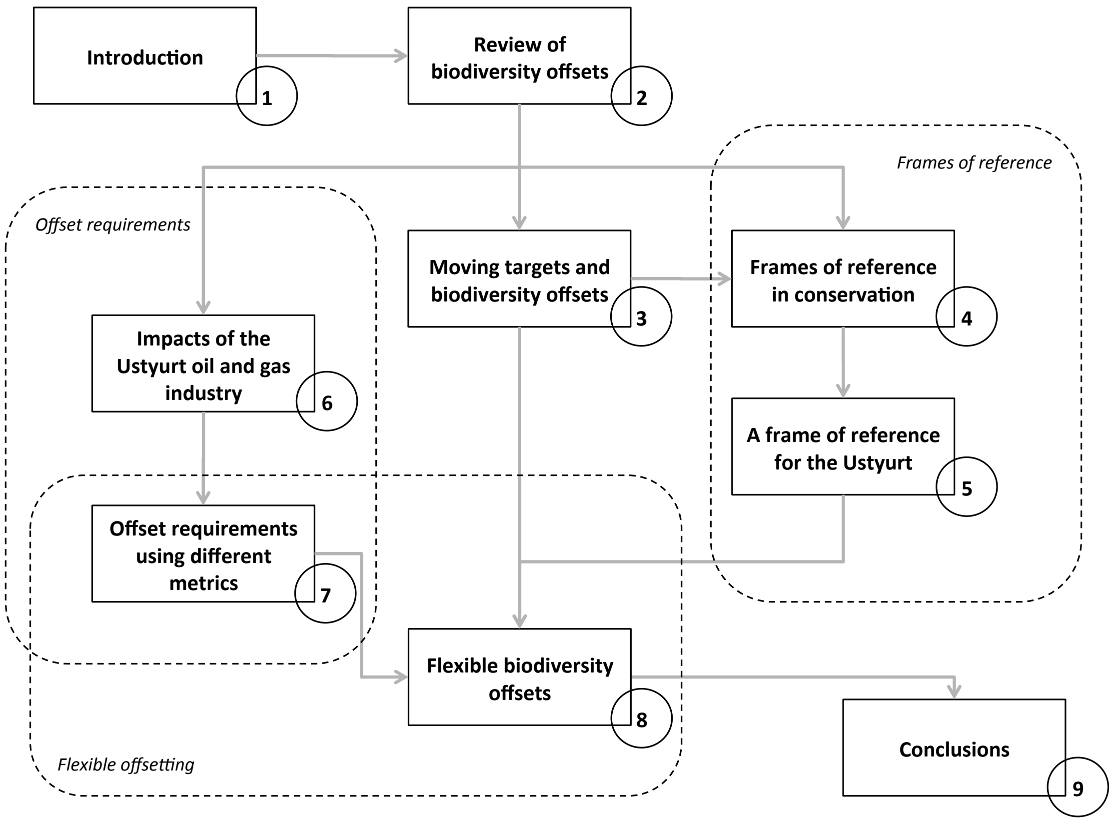
1. to elucidate the challenges facing the offset approach, in general and in the context of social and environmental change;
2. to evaluate key environmental and anthropogenic trends in the Ustyurt plateau, and in doing so better understand the development of frames of reference in conservation;
3. to explore specific challenges associated with biodiversity offsets, particularly the non-static determination of biodiversity losses and required gains;
4. to explore whether, and how, flexible offset mechanisms can deliver long term conservation benefits in the context of socio-ecological change; and,
5. to provide policy advice to those designing biodiversity offset policies based upon the research outcomes.

1.3 Thesis outline

Three research themes run through this thesis, combining exploration of both the theoretical basis for biodiversity offsetting and the broader moving target problem (Fig. 1.1). The first is how to specify an appropriate frame of reference for evaluating no net loss, given that ecosystems are not static (Ferraro & Pattanayak, 2006; Gordon et al., 2011a). The second is how to determine the magnitude of ecological gains required to fully compensate for given losses, which necessarily involves investigation of metrics and uncertainties (Moilanen et al., 2009; Quétier & Lavorel, 2012). The third is the degree to which biodiversity losses and gains traded in offset schemes should be permitted to be out of kind, i.e. how flexible can offsets be (Habib et al., 2013).

This thesis is structured as follows:

***Figure 1.1**: The structure of this thesis by numbered chapter, showing logical flow between chapters, and grouping chapters into the three main themes of the research (frames of reference, offset requirements, and flexible offsets).*



Chapter 2 constitutes a comprehensive review of the literature surrounding biodiversity offsetting as a global conservation mechanism. I focus in particular on detailing the theoretical and practical barriers that exist to successful and effective offsetting. The implementation record to date is compiled as far as possible, and a framework for designing and implementing future offsets is suggested.

Chapter 3 is a conceptual analysis, which outlines the need to use more dynamic approaches to the conservation of biodiversity in a changing world. I propose that biodiversity offsets offer a useful platform for testing such approaches. This chapter also introduces the main case study used throughout the thesis – biodiversity offsets for the oil and gas sector in northwest Uzbekistan.

Chapter 4 compares the outcomes of two very different simulation models, in predicting whether no net loss of biodiversity can feasibly be achieved when evaluating biodiversity offsets against different baselines and counterfactuals. One model is a simple idealized analytical model developed for this purpose; the other is a complex, realistic, stochastic, spatial model of native grassland offsets in Victoria, Australia. I show how the achievement of no net loss in different change scenarios depends on the choice of counterfactual, with implications for the design and subsequent evaluation of offset policy.

Chapter 5 represents an attempt to understand the historical context, establish a project baseline and develop a projected counterfactual for conservation interventions in the Ustyurt plateau, i.e. develop an appropriate frame of reference. The chapter contains analyses of key conservation-relevant socio-ecological trends in study region over the last 100 years, using secondary data sets including satellite imagery. I show that a counterfactual can be developed even in the absence of comprehensive data, and how the projection of counterfactuals links to adaptive management.

Chapter 6 presents the findings from a series of expeditions to the Uzbek Ustyurt and subsequent mapping exercises, in which biodiversity losses from the oil and gas sector were estimated. Vegetation surveys were completed, using the line intercept method, in the vicinity of industrial extractive activity. I demonstrate that the presence of industrial activity completely denudes habitat within the infrastructural footprint, but does not significantly affect vegetative habitat nearby. Maps of oil and gas infrastructure across the plateau are presented, allowing impacts from industry to be scaled up to the landscape scale for the purposes of conservation planning.

Chapter 7 uses data collected from the previous chapter, along with calculations based upon data in the literature, to generate condition-area curves for vegetation cover and saiga habitat biodiversity offsets in the Ustyurt. A range of current national offset methodologies are used to estimate offsets commensurate to oil and gas impacts in the Ustyurt over the last 40 years. I demonstrate how currently employed offset methodologies and metrics generate a range of compensation requirements, highlighting the uncertainty in calculating biodiversity losses and gains. It also begins to consider the possibility of flexibility in biodiversity offsetting.

Chapter 8 uses conservation planning software to explore the use and cost-effectiveness of flexible offsets as a conservation intervention in the Ustyurt. I take the data gathered in previous chapters and analyse it using the Zonation software package, under a set of scenarios corresponding to different levels of flexibility in ecological, spatial and temporal constraints on offsetting. An analysis of the optimum combination of flexible and non-flexible offsets is then completed using the RobOff software package. Overall, I find that flexible offsets would potentially perform better in the context of change scenarios than non-flexible offsets, but that a mixture of the two approaches represents an optimal approach in the Ustyurt.

Chapter 9 highlights and discusses the main conclusions reached in the thesis, and suggests avenues for further research.

Chapter 2

Biodiversity offsets in theory and practice

“All theory is gray, my friend. But forever green is the tree of life.”

Johann Wolfgang von Goethe (from “Faust: part 1”, 1808)

“But Mousie, thou art no thy lane,
In proving foresight may be vain:
The best laid schemes o' mice an' men
Gang aft agley,
An' lea'e us nought but grief an' pain,
For promis'd joy!”

Robert Burns (from “To a Mouse”, 1785)

2.1 Introduction

Conservation concerns are currently ineffectively integrated into development and risk being perceived as incompatible with economic growth. Biodiversity offsets offer an approach that links conservation with industry, potentially providing improved ecological outcomes alongside development. At least 45 policies and programmes exist worldwide in which legislation mandates compensatory biodiversity conservation mechanisms (including offsets), with another 27 under development (Madsen et al., 2011). Voluntary offsets meanwhile, although not legally required, offer a number of potential attractions to developers, as discussed in TEEB (2010) and ten Kate et al. (2004). Consequently, there has been a proliferation of voluntary offsets in recent years.

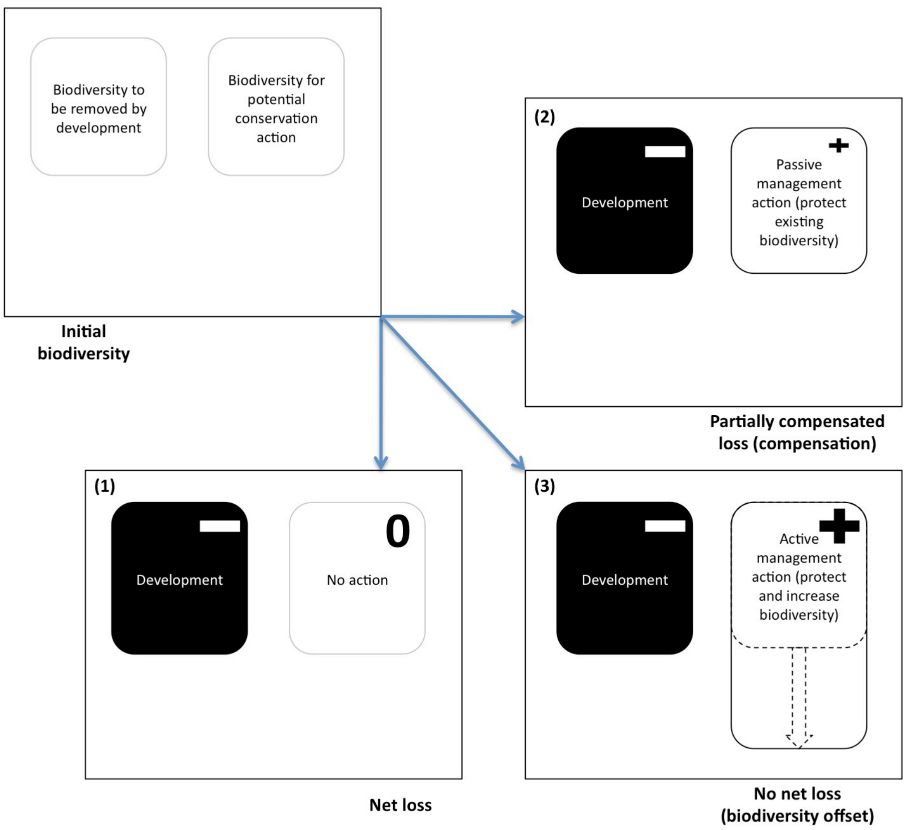
From a conservation perspective, biodiversity offsets may present a conceptually attractive approach (Gibbons & Lindenmayer, 2007; Bekessy et al., 2010). However, substantial problems exist with the perception, design and implementation of offsets. In this review, I first discuss the use of the term 'biodiversity offset', and ambiguities surrounding the way it is defined. I bring together and discuss the disparate theoretical problems identified in the literature that need addressing in order for biodiversity offsets to attain their potential (I define 'theoretical' to mean problems which could in principle be resolved via improved scientific understanding). This leads to a discussion of the practical challenges that have arisen from implementing offset schemes, i.e. those that could be addressed through better governance and existing science. Whilst practical challenges have also been discussed separately in the literature, I bring them together for elaboration, and also combine empirical estimates of implementation success from different national offset policies for the first time. Finally, I propose how these problems could be integrated to allow the development of offset methodologies in a more systematic way.

2.2 What is a biodiversity offset

One definition of biodiversity offsets (‘offsets’) has been created by the Business and Biodiversity Offsets Programme (BBOP), a key actor in the development of offset methodologies. BBOP guidance is widely cited in the literature, providing a useful basis for discussing offsets. However it should be noted that these documents provide one interpretation of biodiversity offsets, and the term offset actually encompasses a range of mechanisms. The BBOP definition states *“Biodiversity offsets are measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken. The goal of biodiversity offsets is to achieve no net loss and preferably a net gain of biodiversity on the ground with respect to species composition, habitat structure, ecosystem function and people’s use and cultural values associated with biodiversity”* (BBOP, 2009a). It should be noted that throughout, unless otherwise stated, the term ‘biodiversity’ is used in the broadest sense (i.e. total biotic variation, from the level of genes to ecosystems).

In line with this definition, offsets are commonly viewed as actions to create additional comparable biodiversity (Fig. 2.1) to compensate for losses caused by development. They are intended as a last resort for developers seeking to compensate for unavoidable damage, after having applied some form of mitigation hierarchy (Kiesecker et al., 2010). This might require, for instance, that developers 'avoid, minimize and rehabilitate' any biodiversity impacts as far as possible, before offsets are then applied to residual impacts (BBOP, 2012). A distinguishing characteristic of biodiversity offsetting is the common inclusion of a 'no net loss' requirement (Fig. 2.1). An alternative interpretation of this stipulation is to say that offset policies require in kind compensation that balances biodiversity losses. Some mechanisms go further, aiming for a 'net gain' in biodiversity. All such outcomes are pursued by quantifying residual ecological impacts arising from development, and creating equivalent biodiversity components elsewhere (BBOP, 2009a). In reality, phrases such as *no net loss* and *in kind compensation* hold different meanings for different stakeholders, and offset schemes consequently vary significantly in their objectives, methodologies and project delivery.

Figure 2.1: *Schematic of the offsetting principle for development impacts. A development that will damage biodiversity is contemplated (top left). Some options are: (1) development only, resulting in net loss (-) of biodiversity; (2) protect existing biodiversity elsewhere, resulting in a compensated net loss (large - and smaller +); or (3) create or restore additional comparable biodiversity elsewhere, resulting in no net loss i.e. a biodiversity offset (- and + same size).*



Offsets is a term given to a family of related policies, also known as ‘compensatory habitat creation’ (Morris et al., 2006), ‘mitigation banks’ (Gibbons & Lindenmayer, 2007) ‘conservation banking’, ‘habitat credit trading’, ‘complementary remediation’ and more (Madsen et al., 2011). Offset ‘banks’ are essentially where providers have created offset project/s in exchange for biodiversity credits, which can subsequently be sold to compensate for developments with comparable residual ecological impacts. It should be noted that the concept of utilizing a banking mechanism for offset schemes is less mature by at least 10 years than the concept of offsets itself (Environmental Law Institute, 2002). The BBOP guidance further characterises offsets as primarily one-off conservation projects tied to a given development, that specifically require: “*measurable conservation gains, deliberately achieved to balance any significant biodiversity losses that cannot be countered by avoiding or minimising impacts from the start, or restoring the damage done*”, and “*no net loss of biodiversity from the perspective of all relevant stakeholders*” (BBOP, 2009a).

Offsets are often considered a market-based instrument for biodiversity, enabling a ‘baseline and credit’ market (eftec et al., 2010; Parker & Cranford, 2010; Wissel & Wätzold, 2010) for the trade of biodiversity “value”. Indeed, systems such as Wetland Banking in the US (US NRC, 2001) and BioBanking in Australia (DECCW, 2009) specifically create markets for biodiversity credits. But the impossibility of defining a consistent, fungible unit that comprehensively captures biodiversity (Purvis & Hector, 2000) means that biodiversity itself is not a tradable market commodity (Walker et al., 2009; Salzman & Ruhl, 2000), hence the need for proxies such as credits. Credits are complicated by the fact that the conservation value of any one component of biodiversity is not fixed, for instance, being dependent upon its spatial relationship with biodiversity elsewhere (Drechsler & Wätzold, 2009). Offsets therefore do not enable a market for biodiversity as readily as they do for pollution (Godden & Vernon, 2003). Rather, offsets are effectively a mechanism for pricing certain negative environmental externalities into development projects.

Three criteria can be distilled, common to key legal offset policies (McKenney & Kiesecker, 2010) and BBOP guidance, which in combination make offsets unique:

1. Provide additional **substitution** or **replacement** for **unavoidable** negative impacts of human activity on **biodiversity**;
2. Involve **measurable**, **comparable** biodiversity **losses** and **gains**; and,
3. **Demonstrably** achieve, as a minimum, **no net loss** of biodiversity.

I use these criteria to define offsets for the remainder of this thesis, believing them to be consistent, in principle, with the majority of offset schemes. There is value to conceptualizing offsets into consistent, comparable terms such as these. If schemes were compared under a common conceptual framework, this would provide a more effective means for scientific evaluation and development of best practice methodologies. This would enable rigorous comparison of strengths and weaknesses across offset programs, and allow ongoing improvement to schemes. The criteria can result in a variety of different interpretations and assumptions at the implementation level. It is from these criteria that a host of theoretical and practical challenges arise, which are the subject of this review.

2.3 Challenges for biodiversity offsets in theory

The three criteria outlined in the definition above engender a number of questions, including the necessity to define biodiversity and choose a metric for measuring it. I summarise my view of these unresolved theoretical problems associated with offsets (Table 2.1), whilst expanding upon each problem below and making management recommendations.

***Table 2.1**: A summary of key theoretical challenges, with design recommendations, for biodiversity offsets*

Problem	Description	Relevant research	Design recommendations
(a) Currency	Choosing metrics for measuring biodiversity	McKenney & Kiesecker (2010); Temple et al., (2010); Treweek et al. (2010); BBOP (2009a); Norton (2009); Walker et al. (2009); Burgin (2008); Chapman & LeJeune (2007); McCarthy et al., (2004); Godden & Vernon (2003); Salzman & Ruhl (2000); Humphries et al. (1995)	Use multiple or compound metrics Incorporate measure of ecological function as well as biodiversity
(b) No net loss	Defining requirements for demonstrating no net loss of biodiversity	Maron et al. (2013); Gordon et al. (2011a); Bekessey et al. (2010); McKenney & Kiesecker (2010); BBOP (2009a); Gorrod & Keith (2009); Gibbons & Lindenmayer (2007)	Measure no net loss against dynamic baseline, incorporating trends State whether no net loss is at project or landscape level. Consider discounting rate (e.g. Dunford et al., 2004)
(c) Equivalence	Demonstrating equivalence between biodiversity losses and gains	Habib et al. (2013); Quetier & Lavorel (2011); Burrows et al. (2011); McKenneny & Kiesecker (2010); Bruggeman et al (2009, 2005); Norton (2009); Chapman & LeJeune (2007); Gibbons & Lindenmayer (2007); Godden & Vernon (2003)	Do not allow 'out of kind' trading *unless* ''trading up' from losses that have little or no conservation value
(d) Longevity	Defining how long offset schemes should endure	Overton et al. (2013); Pouzols et al. (2012); McKenneny & Kiesecker (2010); BBOP (2009a); Gibbons & Lindenmayer (2007); Morris et al. (2006)	Offsets should last at least as long as the impacts of development Offsets should be adaptively managed for change
(e) Time lag	Deciding whether to allow a temporal gap between development and offset gains	Maron et al. (2012); Drechsler & Hartig (2011); Gordon et al. (2011a); Bekessey et al. (2010); McKenneny & Kiesecker (2010); Moilanen et al. (2009); Norton (2009); Gibbons & Lindenmayer (2007); Morris et al. (2006)	Require offsets to be delivered through biodiversity banking mechanisms
(f) Uncertainty	Managing for uncertainties throughout the offset process	Pickett et al. (2013); Maron et al. (2012); Pouzols et al. (2012); Treweek et al. (2010); Moilanen et al. (2009); Norton (2009)	Development of a framework for uncertainty in offsets is a research requirement
(h) Reversibility	Defining how reversible development impacts must be	Maron et al. (2012); BBOP (2012); Godden & Vernon (2003)	Define reversibility Require all biodiversity losses to be reversible
(i) Thresholds	Defining threshold biodiversity values beyond which offsets are not acceptable	Pilgrim et al. (2013); BBOP (2012); BBOP (2009a); Norton (2009); Gibbons & Lindenmayer (2007); Morris et al. (2006)	Define explicit thresholds for non-offsettable impacts

(a) Currency

There exists no single metric that objectively captures the full extent of biodiversity, which itself has no universal, unambiguous definition. Any measure of biodiversity is therefore a proxy (Humphries et al., 1995). However, offsets ostensibly rely upon the accurate quantification of losses and gains, and therefore require robust metrics (Burgin, 2008).

Various metrics are used in offsets. The use of single metrics like ‘area of habitat’ to represent biodiversity losses and gains has been widely discredited (TEEB, 2010). Compound metrics can be used e.g. Victorian (Australia) offset schemes, where the basic currency is a composite metric Habitat Hectares (HH) (DSE, 2002). The HH score summarises information about an area including the relative condition of the vegetation and its spatial context within the landscape, making it useful for management (McCarthy et al., 2004), but does not capture information about other elements of biodiversity e.g. genetic diversity. The use of multiple metrics may result in a more comprehensive understanding of biodiversity losses and gains (e.g. Kiesecker et al., 2009), and all offset designers should be increasingly expected to employ multiple or compound metrics.

An important question is whether offsets should be intended to provide compensation for biodiversity, ecosystem function, ecosystem services, or all three. Different stakeholders might desire no net loss of diversity (e.g. Australian grasslands), ecosystem function (e.g. US wetlands), or the provision of services such as carbon sequestration. US conservation banking focuses on species diversity, but Bruggeman et al. (2005, 2009) explore how function and genetic diversity could be used as alternatives. The topic of measuring diversity alongside function is the subject of ongoing research (Cadotte et al., 2011), and arguably offsets should not target diversity alone.

(b) No net loss

The requirements for demonstrably achieving no net loss are often undefined. In particular, the baseline against which to measure no net loss is rarely specified. The implicit assumption is often that the biodiversity baseline is fixed at the point of development. However, as ecosystems are dynamic, no net loss should be defined against prevailing trends in biodiversity condition. For example, native Australian grassland is deteriorating due to aggressive invasive species, so managing grassland to prevent further degradation could deliver a gain against a baseline that incorporated predicted landscape trends (Gordon et al., 2011a). Thus, if clearing grassland for development, an offset maintaining current grassland condition in other areas could be said to deliver no net loss. This is a different form of additionality to active habitat creation, e.g. creation of new wetlands under US wetland banking (US EPA, 2008), which results in no net loss against a fixed baseline. It is recommended that, as for European EIA legislation (eftec, 2010) no net loss be defined against a dynamic baseline that incorporates trends.

It is not clear how best to aggregate biodiversity losses and gains across a landscape, and thereby judge the efficacy of an offset policy. No net loss could be measured against change at project level, or across the wider landscape. For example, in the New South Wales landscape, there has been an absolute net loss in native vegetation since the 2005 Vegetation Act (Gibbons & Lindenmayer, 2007). But in the same time period, a reduction in approvals for vegetation clearance was achieved (Gibbons, 2010), which was a net gain against a business as usual scenario at the project level. Despite an increasing emphasis on landscape-scale outcomes in conservation, in general, when the policy objective is no net loss this implicitly means no net loss at project level. It is important that the scale used in a given offset scheme is made explicit. Concurrently, the possibility for 'leakage' of development impacts outside of the area evaluated under the offset policy should be considered.

The trading of biodiversity losses against gains in a dynamic system should include the application of a discount rate (Moilanen et al., 2009). The application of discount rates enables appropriate trading of future gains against current losses (Dunford et al., 2004). Whilst often ignored, some approaches do incorporate time discounting (Pouzols et al., 2012).

(c) Equivalence

It is difficult to argue ecological equivalence between biodiversity components that differ in type, location, time or ecological context. This is the case even for 'like' habitat, for instance, a manmade wetland is demonstrably not equivalent to a naturally established wetland (Moreno-Mateos et al., 2012), as implied under US Wetland Banking.

It becomes more difficult when trading 'out of kind', e.g. trading losses of adult seabirds from fishing bycatch for gains in nesting habitat (Wilcox & Donlan, 2007), or trading losses for spatially distant gains. Different biodiversity components are traded under some schemes, e.g. habitat types are exchangeable in the UK (Defra, 2011). The fact that biodiversity is not fungible calls into question the use of out of kind offsets (Godden & Vernon, 2003; Salzman & Ruhl, 2000). But by allowing this, 'trading up' is possible i.e. trading losses in habitat of low conservation significance for gains in threatened habitats (Quétier & Lavorel, 2012). The latter case is one in which there is a compelling argument for out of kind trades.

Importantly, recreating biodiversity does not necessarily result in restoring the functional performance of the previously existing system (Ambrose, 2000). This reinforces that offsets should incorporate some measure of function alongside diversity.

(d) Longevity

There are two distinct longevity challenges: firstly, defining how long offsets are expected to last i.e. the time horizon for evaluation. Secondly, ensuring offsets are designed to endure for that long in a dynamic environment.

Concerning expected longevity, offsets might be required to last for as long as the impacts of development, or in perpetuity (BBOP 2009a). ‘In perpetuity’ is not necessarily considered ‘forever’: e.g. the REMEDE toolkit (Lipton et al., 2008) approximates 'in perpetuity' to 50–75 years based on a positive discount rate. Given difficulties in agreeing ‘in perpetuity’, let alone the management implications, requiring offsets to last as long as development impacts seems a more practical goal. This would however to a degree rest upon development impacts being reversible (see h.) against the project baseline, and therefore depends ultimately on other elements of an offset policy.

The long-term persistence of any offset project might be threatened by environmental change (Chapter 4). Of course, the same change may have applied to the original habitat for which the offset was created. How best to account for and incorporate change is a theoretical challenge. But additionally, it is not always clear how an offset should be maintained, by whom, and for how long. Addressing these issues becomes a problem of implementation, and is vital if a scheme is to achieve long term no net loss.

(e) Time lag

There can be a temporal gap between development impacts occurring and the benefits associated with the offset scheme accruing. Therefore, whilst biodiversity losses are guaranteed, future gains may be realized late or not at all e.g. the condition of restored habitat is increasingly uncertain further into the future (Bekessy et al., 2010). For instance, due to the time associated with grassland restoration, offset schemes in Victoria can result in temporary losses in total grassland condition across the landscape, as measured in HH (Gordon et al., 2011a). Alternatively, the political or legal landscape can change at any time, as for the Brazilian Forest Code (Madsen et al., 2011).

Time lags interact with fluctuations in biodiversity credit prices to result in reduced efficiency in biodiversity markets (Drechsler & Hartig, 2011). Further, interim losses of biodiversity may be unacceptable: either because they have detrimental impacts upon the wider ecosystem, or because they represent a temporary lack of ecosystem service provision. Solutions to this include requiring offsets to be implemented before development (Bekessey et al., 2010), or applying time discount rates.

(f) Uncertainty

The outcomes of offset schemes are subject to uncertainty. This is often accounted for simplistically by increasing the amount of compensation required i.e. using multipliers. A multiplier increases the amount of biodiversity gains required based on various factors, such as theoretical uncertainty in the definition and measurement of biodiversity, and the need for a discount rate for future gains. The largest obligatory multipliers come under the Western Cape offset policy, requiring compensation of 30 ha of land for every hectare cleared in critically endangered habitats (DEADP, 2007). Arbitrary multipliers like this take a risk averse approach, but may be insufficient to address correlated losses or total failure of an offset scheme (Moilanen et al., 2009). Although investigation into managing uncertainty in offsets continues (e.g. Pouzols et al., 2012), the development of a comprehensive framework for treating uncertainty in offsets is an important research requirement.

(h) Reversibility

The impacts of development on biodiversity could in some situations be reversed through restoration. For instance, clearance of shrubby vegetation by vehicles for gas exploration in semi-arid regions in Uzbekistan is generally thought reversible in the short term through restoration (Chapter 6). However, if the same exploration activity created roads that facilitated access for poachers to extirpate an endangered species in previously inaccessible areas, this might prove irreversible.

Reversibility is considered a prerequisite for the viability of offsets as a general policy tool (Godden & Vernon, 2003), so ideally all biodiversity losses addressed through offsets should be reversible. However, reversibility has no objective definition, and policy must define it explicitly. An example of a policy containing comment on this is that of the Western Cape, which specifies that “ecosystems that have undergone severe degradation of ecological structure, function or composition as a result of human intervention and are subject to an extremely high risk of irreversible transformation” are non-offsettable (DEADP, 2007).

(i) Thresholds

Defining thresholds, beyond which the use of offsets is considered inappropriate, involves making value judgments. In this context, thresholds are considered a type or magnitude of loss beyond which impacts are deemed non-offsettable. For example, species extirpation could be considered unacceptable, and therefore non-offsettable, whereas temporary grassland impacts might be deemed acceptable. Consequently, it is difficult to create protocols for setting thresholds (BBOP, 2012). Society might accept a scheme that treats some habitat types as interchangeable, as in UK offsets (Defra, 2011). However, the same scheme might not be acceptable if it involves the loss of charismatic fauna: Donlan & Wilcox (2007) explore the possibility of offsetting seabird bycatch in fisheries, provoking heated debate. The explicit definition of thresholds is therefore fundamental to offset design.

2.4 Challenges for biodiversity offsets in practice

The theoretical hurdles to offsets are compounded by practical challenges, which I broadly group into three categories: compliance, measuring ecological outcomes, and uncertainty (Table 2.2). The implementation record for offsets to date is limited and confined to developed countries

(Table 2.3), but offsets are increasingly being explored in the developing world, where the issues of implementation may be even more acute.

***Table 2.2**: A summary of practical challenges for the offset approach. Examples discussed within the main text.*

Root problem	Result	Example
(1) Compliance	• Non-compliance with the mitigation hierarchy	Mühlenburger Loch, Germany
	• Insufficient compensation proposed	Mühlenburger Loch, Germany
	• Offsets not implemented, or only partially implemented	Wetland banking, US Fish habitat, Canada
	• Legislation changes during offset scheme	Forest Code, Brazil
(2) Measuring ecological outcomes	• Monitoring different things suggests different ecological outcomes	Wetland banking, US
	• Difference in opinion about ecological outcomes	Basslink project, Australia
	• Outcomes not measured for very long	Fish habitat, Canada
	• Outcomes not monitored	Conservation banking, US
	• No follow up by regulator	Conservation banking, US
(3) Uncertainty	• In measurement of biodiversity baseline	Native grassland, Australia
	• In magnitude and type of development impacts	Extractive sector, Uzbekistan
	• Offsets fail to establish or persist	Wetland banking, US
	• Development causes greater impacts than expected	Fish habitat, Canada

Table 2.3*: Implementation record for biodiversity offsets in Canada, the US and Australia.*

Related to	Country	Mechanism	Implementation success rates		Sample size	Reference
Compliance, Uncertainty	US	Wetland banking	30 %	of offsets meet all project objectives	76 sites	*Matthews & Endress (2008)*
	US	Wetland banking	50 %	of offsets fully implemented	23 sites	*Mitsch & Wilson (1996)*
	US	Wetland banking	74 %	of offsets achieve no net loss	68 banks	*Brown & Lant (1996)*
	Canada	Fish habitat compensation	12 - 13 %	of offsets implemented as required	52 sites	*Quigly & Harper (2006a)*
Monitoring ecological outcomes	Australia	Native vegetation compensation	80 %	reduction in approvals for vegetation clearance	Across New South Wales, Australia	*Gibbons (2010)*
	US	Wetland banking	0 %	of created wetlands were functionally successful	40 sites	*Sudol (1996) in Ambrose (2000)*
	Canada	Fish habitat compensation	37 %	of offsets didn't result in a loss of productivity	16 sites	*Quigly & Harper (2006b)*

(1) Compliance

Noncompliance with offset requirements is a significant challenge, and takes a variety of forms (Table 2.2). Developers may not comply with the mitigation hierarchy: for instance, a proposed development in Germany involves impacts upon Mühlenburger Loch, a protected area. Planning permission was applied for on the grounds of 'no alternative sites', with proposals for compensation. The EU Commission placed the case under examination, concluding that the developer had not sufficiently considered alternative sites (Kramer, 2009).

The Mühlenburger Loch case also provides an example of a developer not proposing sufficient compensation. The proposals entailed replacing approximately 170 ha of wetland with comparable habitat across 4 sites. However the proposals would have resulted in 100 ha of comparable habitat (Pritchard et al., 2001). This problem could arise through a lack of clarity in defining no net loss and equivalence, or poor practice by the developer.

Alternatively, noncompliance can lead to offset projects being implemented partially or not at all. This outcome has certainly occurred in the case of US wetlands (Mack & Micacchion, 2006; Matthews & Endress, 2008). Effective wetland banking is consequently considered achievable in principle, but not yet achieved consistently in practice. This is similarly the case for Canadian fish habitat compensation (Quigly & Harper, 2006a), and the Brazilian Forest Code (Hirakuri, 2003).

Revision of legislation after compensation schemes have begun further complicates the issue of legal compliance. An example is the Brazilian Forest Code, which allows trade in forest set-asides (McKenney & Kiesecker, 2010) and is currently undergoing significant amendment (Madsen et al., 2011).

(2) Measuring ecological outcomes

There is limited quantitative information available on the outcomes of offset projects. This is consistent with the broader issue of the lack of post-implementation evaluation in conservation (Ferraro & Pattanayak, 2006).

Even if offsets are monitored, it is not necessarily clear whether the ecological outcomes have been positive. Confer & Niering (1992) recorded comparable biodiversity across created and natural US wetlands, but higher floral species richness in created sites, versus increased wildlife sightings and less invasive species at natural sites. This is partially associated with the lack of a comprehensive biodiversity currency.

Similarly, different parties might evaluate project success differently, dependent upon motivation, analytical techniques, or methodology. The ‘Basslink’ marine pipeline project in Australia (Westerweller & Price, 2006) resulted in impacts that were managed for net gain in native vegetation, and some treat the project as successful (BBOP, 2009b). But other studies conclude that overall impact was negative, with offsets not achieving project objectives (Duncan & Hay, 2007).

A lack of robust information on outcomes may equally be due to those responsible failing to monitor offsets adequately. In Canada, offsets were only monitored for an average 3.7 years post construction (Harper & Quigly, 2005), and it is unknown whether Canadian compensation policy objectives were achieved (Rubec & Hanson, 2009). In the US, similar conclusions were reached for Conservation Banking after a decade of implementation (Carroll et al., 2008).

Rigorous post-implementation evaluation is the only way to know whether losses and gains are balanced in the long term, and no net loss ensured. Equally, a track record of successful implementation is necessary to demonstrate that offsets can work in practice. Currently there is no publically available global register of offset project outcomes: this would aid understanding of the long-term effectiveness of offsets.

In part, this issue relates to responsibility and the burden of proof. It is not always clear who is responsible for delivery of offsets, during implementation and into the future. Uncertainty over the burden of proof could be avoided if responsibilities throughout the full project lifecycle were defined from project inception.

(3) Uncertainty

A concern for conservation is to ensure that interventions incorporate consideration of uncertainty (Hilderbrand et al., 2005; Langford et al., 2009). Uncertainty arises at every stage of offsetting and the lack of any sophisticated applied treatment of uncertainty is a major shortfall, although the “RobOff” software developed by Pouzols et al. (2012) begins to address this need. I utilise the taxonomy of uncertainty developed by Regan et al. (2002; Table 2.4) to give an example of how such a framework might begin to be structured. Note that this is only intended as one example of a possible framework for dealing with uncertainty, and does not necessarily cover every conceivable uncertainty; for instance, see Kujala et al. (2012).

***Table 2.4**: Example of a structured classification of uncertainty in offsets, using the taxonomy from Regan et al. (2002)*

Category	Source	Example: uncertainty in offsets
Epistemical	Measurement	Error in measuring biodiversity losses and gains
	Systematic uncertainty	Excluding unknown biodiversity when measuring losses
	Natural variation	Habitat restoration not guaranteed to succeed
	Inherent randomness	Unpredictable events e.g. extreme weather affect offset
	Model uncertainty	Error in projections of habitat impacts from climate trends
	Subjective judgement	Error in estimating total species abundance from available data
Linguistic	Vagueness	Including endangered species in offsets. The word ‘endangered’ can be vague
	Context dependence	Defining ‘high biodiversity’. Could mean high species richness, high endemicity, high uniqueness, etc
	Ambiguity	‘No net loss’ can have different meanings against different baselines
	Under-specificity	Insufficient ecological information provided on development impacts to calculate true losses
	Indeterminancy in theoretical terms	The classification of habitats changing with time

Uncertainty in offset implementation is widely managed through multipliers (see f. above) or via conservation banking (Bekessy et al., 2010). Information is often insufficient to generate realistic multipliers. There may also be insufficient motivation to use the multipliers necessary for ‘robustly fair’ offsets for practical or financial reasons, if as large as those derived by Moilanen et al. (2009).

There may be uncertainty around whether the offset provider possesses competence to establish successful offsets, or whether sufficient land exists in an area to provide offsets for all developers who require it. Finally, the future gains from offsets contain significant uncertainties. Restored or created habitats might fail to establish or provide sufficient ecological function (Ambrose, 2000), or impacts may be greater and compensation less than planned (Table 2.3). This is a combination of ecological uncertainty and uncertainty in the actions of developers and offset providers.

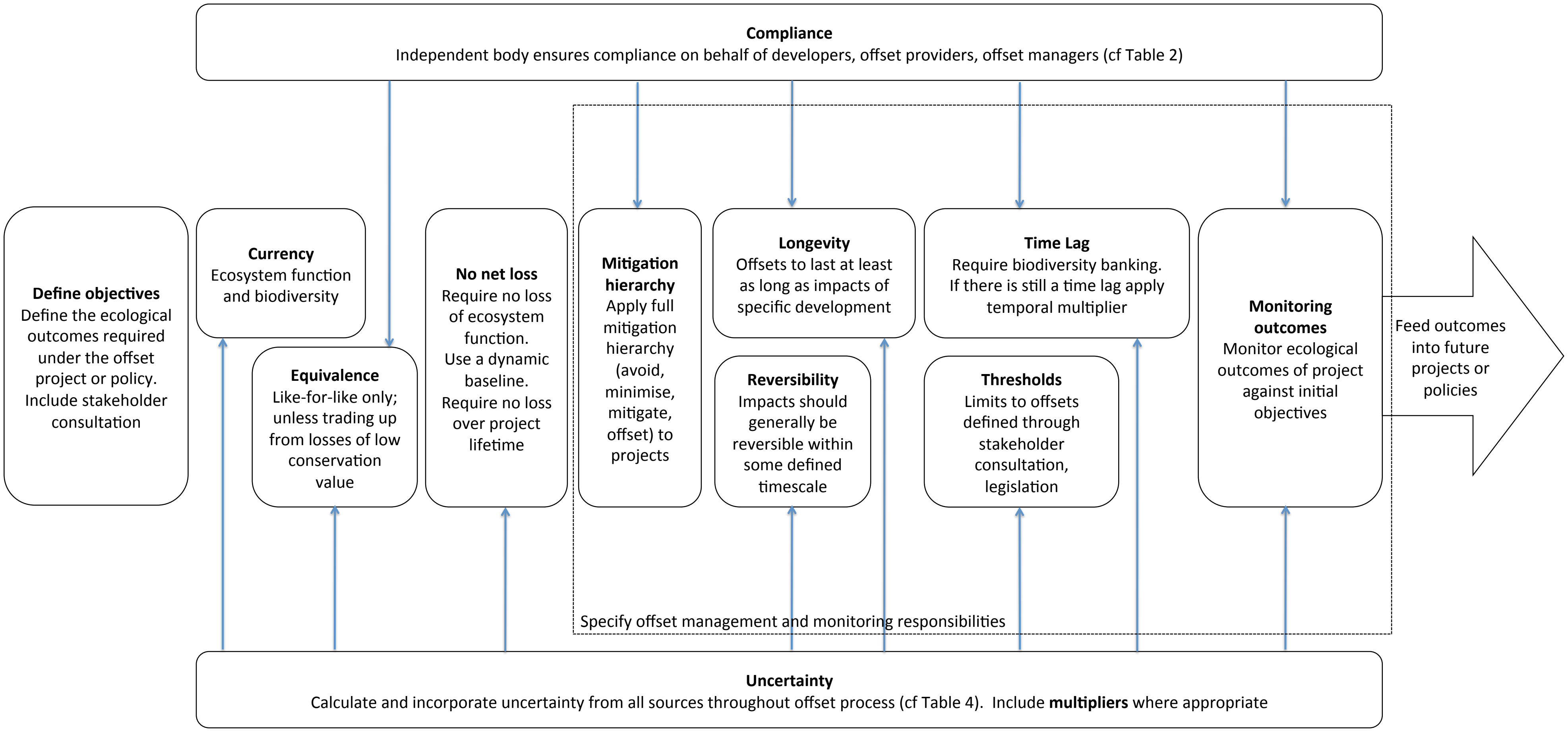
2.5 Management of theoretical and practical challenges

Offsets are faced with both theoretical (Tables 2.1, 2.4) and practical (Tables 2.2, 2.3, 2.4) challenges. Research could focus on resolving theoretical problems and developing universally applicable principles for offset design. Equally, researchers could concentrate upon monitoring and evaluating implementation for various approaches, each designed to resolve locally specific practicalities. Whilst BBOP (2012) pursues the former path, the field is in practice moving towards the latter.

These challenges are intertwined and must be resolved in conjunction. For instance, the problem of choosing a metric for biodiversity complicates the problem of defining no net loss and equivalence. The definition of no net loss begs questions about the required longevity of the offset and the acceptability of time lags. If time lags are permitted, then ensuring offsets are delivered at all becomes a practical challenge, in terms of uncertainty in offset outcomes and in ensuring that offset providers deliver those outcomes. Subsequently, if ecological outcomes are not monitored then it is difficult to demonstrate no net loss or improve knowledge on appropriate multipliers and thresholds.

A basic conceptual framework that integrates all of these problems could facilitate a common approach to managing the challenges associated with offsets, and allow systematic development of the offset methodology. I propose such a framework (Fig. 2.2), which could be used to contrast and improve methodologies.

***Figure 2.2**: Conceptual framework for integrating theoretical and practical problems in offsets. Process reads from left to right.*



2.6 Discussion

This review highlights the myriad conceptual and practical challenges that beset offsets. In some areas management recommendations can be made, in others there are aspects of offsetting that require further research. I conclude with a discussion based around three issues I believe need particularly urgent research to ensure robust offsetting.

2.6.1 *Biodiversity, function or services?*

The stated intention of offsets is commonly to ensure no net loss of biodiversity. However no net loss can also mean no loss in ecosystem function, or in the value provided to society by functioning biodiversity, i.e. ecosystem services. Biodiversity has been suggested to underpin ecosystem function (CBD 2010) or be closely related to it (Nelson et al., 2009). However high biodiversity does not necessarily coincide with provision of particular ecosystem services (Naidoo et al., 2008). Consequently, an offset scheme could be targeted at retaining biodiversity, function, services or all three; but these are not always compatible goals. More research is required to determine where it is possible to conserve all three simultaneously (Cadotte et al., 2011). It is important that offset schemes are clear about which they aim to conserve.

2.6.2 *Dynamic baselines and multiple metrics*

Offsets could use fixed or dynamic baselines against which to measure no net loss. The latter would account for e.g. drivers such as climate change (Chapter 4). However, predicting future biodiversity trajectories accurately is difficult, and managing for them perhaps impractical. Further, overcomplicating the theoretical basis of offsets in this way may also risk undermining one of the key benefits to the approach: the flexibility and perceived simplicity that appeals to business and policymakers. Consequently, research that explores how to specify dynamic baselines, and what conservation actions would be required under different baselines, would be useful.

Similar arguments apply to the use of multiple metrics. Additional metrics result in additional complication and expense, and beyond some point will not justifiably further reduce uncertainty in quantifying biodiversity. Therefore, exploration of how to optimise the use of multiple metrics for offsets is necessary.

2.6.3 *Implementation of offsets in the developing world*

Offsets have been used since the 1970s, but at best have been only modestly successful (Table 2.3). The issues facing the implementation of offsets in highly industrialised nations could potentially be magnified in developing countries (ten Kate et al., 2004). Here, global conservation priorities might co-exist with intense natural resource use, and there may be differences in the language of legislation, policy and expertise. Designers of voluntary offsets in developing countries may need to incorporate different perspectives on, or a lack of: environmental legislation and Environmental Impact Assessment; environmental management; policy or regulatory frameworks; information on biodiversity, indicators and threats; monitoring capability or funding; enforcement resources or infrastructure; and, local technical expertise or capacity. This applies equally to the creation of biodiversity markets through tradable credits, which necessitate some of these elements (Wissel & Wätzold, 2010).

The use of offsets in emerging economies is by no means impossible or even inadvisable. A number of countries currently host offset pilots e.g. Uzbekistan, or have related legislation e.g. Brazil (UNDP 2010a; Madsen et al., 2011). Equally, offsets offer collateral benefits that could be magnified in developing nations. These include promoting stakeholder engagement in conservation, leveraging funding to meet strategic conservation objectives and catalyzing improvements in environmental legislation. They could also increase baseline ecological knowledge and expand scientific capacity.

Benefits aside, the potential exacerbation of the issues discussed in this review should be incorporated into the design process when considering offsets in more challenging contexts. Under some conditions offsets may simply not be appropriate. Examples of these conditions include the absence of transparent value judgments made in relation to the theoretical challenges discussed, overwhelming ecological uncertainty, and those cases in which compliance cannot be assured.

Chapter 3

Conservation when nothing stands still

"The surface of a slate-grey lake is lit
By the earthed lightening of flock of swans,
Their feathers roughed and ruffling, white on white,
Their fully-grown headstrong-looking heads
Tucked or cresting or busy underwater.
Useless to think you'll park or capture it
More thoroughly. You are neither here nor there,
A hurry through which known and strange things pass."

Seamus Heaney (from "Postscript", 1996)

"If you try to change it, you will ruin it. Try to hold it, and you will lose it."

Lao Tzu (from the "Tao Te Ching", 6th Century BC)

3.1 Introduction

Conservation of biodiversity relies heavily upon the use of fixed protected areas ('PAs'). PAs are considered 'fixed' if they are stationary on the landscape or seascape and therefore static in space and time (Rayfield et al., 2008). But when conservation targets are not also stationary on the same scale – which is common – the effectiveness of PAs may be compromised. For instance, migratory or nomadic species, which might regularly move through a larger area than is feasible to designate as a PA, can be poorly served by static PAs over much of their life cycle (Singh & Milner-Gulland, 2011). Habitats themselves 'move' over longer timescales, driven by environmental change and anthropogenic activities. To be effective, it is increasingly recognized that conservation planning must account for the dynamic nature of ecosystems (Nicholson et al., 2009), and do so at large (e.g. landscape) spatial scales.

Dynamic conservation problems will require dynamic new conservation approaches. There have been various suggestions as to what more dynamic conservation might look like, but these proposals have yet to be widely tested in practice (Game et al., 2009; Milner-Gulland et al., 2011). This is a crucial challenge for today's conservationists. In this Chapter, I propose one way of possibly addressing this challenge: through the implementation of biodiversity offsets in a way that incorporates consideration of the moving target problem. Biodiversity offsets potentially provide the necessary framework under which to test more dynamic conservation approaches; and here I begin to explore how this might work, using the aforementioned Uzbek case study (Chapter 1).

As discussed in the previous Chapter, the aim of biodiversity offsets ('offsets') is to make developers fully compensate for any biodiversity losses associated with development, with a common aim being to demonstrate 'no net loss' (NNL) of biodiversity overall. The approach originates in the US Water Resources Act of the 1970s, where development resulting in unavoidable destruction of wetland habitat is compensated for via creation of equivalent wetland elsewhere. Whilst this effectively results in fixed PAs, offsets can and are delivered through a range of alternative conservation actions. The NNL requirement means that it is necessary to specify a project baseline against which to measure biodiversity 'losses' and 'gains' (Gordon et al., 2011a). It has additionally been argued, and in some countries is the case, that NNL must be defined not only in terms of space, e.g. by maintaining the overall area of a given habitat; but also in time, e.g. by having to establish biodiversity gains before development proceeds (Bekessy et al., 2010). Demonstrating NNL in space and time should really be a minimum objective of any conservation intervention, but offsets make this requirement explicit. This forces implementers to specify baselines and evaluate future conservation outcomes; in turn meaning they must actively consider the uncertain and dynamic nature of conservation targets.

Uncertainty is partly taken into account within offset schemes through the use of spatial and temporal multipliers (i.e. the amount of offsetting required per unit of projected biodiversity loss is multiplied by a factor to account for uncertainty surrounding that projection; Moilanen et al., 2009). However there is a difference between uncertainty arising from inherent ecological dynamics or lack of knowledge (which can be addressed using multipliers), and the variation in ecological outcomes caused by projected external influences, such as human activities causing habitat loss or climate change. To achieve NNL, by implication, offsets should consequently be designed so that biodiversity gains are maintained in the face of external social and environmental trends. This challenging requirement sets offsets apart from more traditional approaches. Offsets therefore not only offer us an opportunity to test dynamic approaches to conservation; they *must* incorporate such approaches.

In recent years, offsets have attracted the interest of businesses, governments and NGOs, and are rapidly being implemented worldwide: from native grassland in Australia, to fish habitat in Canada, 'Natura 2000' sites in the EU, rainforest in Brazil, and even animal species in the US (McKenney & Kiesecker, 2010). Ensuring their effectiveness in an age of change is consequently of significant importance to academics, practical conservationists, and policy-makers.

3.2 Migratory species as a classic case of moving targets

Migratory species provide a classic example of the challenges that moving targets present for conservation. It is not merely the case that many migratory species are endangered; more fundamentally, large-scale migration *as a phenomenon* is under threat of extinction (Harris et al., 2009).

Many species migrate over such a distance that protecting their entire range is unfeasible; due to costs, competing land uses, or complications around geopolitical boundaries (Milner-Gulland et al., 2011). Fixed PAs can sometimes prove beneficial for a migratory population despite not covering its full range, as observed for wildebeest *Connochaetes taurinus* in the Serengeti (Thirgood et al., 2004). But PAs cannot generally be relied upon to conserve a transient species if planned without taking into account how populations are connected across their lifecycles. Martin et al. (2007) model the migratory American redstart *Setophaga ruticilla*, predicting that partially protecting its range could conserve the species, but only so long as the animal's requirements throughout its range are considered. A similar prediction has been made for certain mobile fish species (Apostalaki et al., 2002).

As an alternative to fixed PAs, 'mobile' PAs could prove valuable in marine conservation (Game et al., 2009) and for migratory species conservation in general (Milner-Gulland et al., 2011). Mobile PAs move with the target species itself, or temporarily coincide with a vulnerable stage in its lifecycle. The Soviet Union once implemented mobile PAs to track calving locations of the migratory saiga antelope *Saiga tatarica* in Kazakhstan (Robinson et al., 2008); but otherwise, the idea is mainly hypothetical. Mobile PAs have been proposed that change location annually for Canadian caribou *Rangifer tarandus* (Taillon et al., 2012) and in real time for southern blue-fin tuna *Thunnus maccoyii* off Australia (Hobday & Hartmann, 2006). Shillinger et al. (2008) propose temporary marine PAs covering annual migration corridors for leatherback turtles *Dermochelys coriacea*. Crucially, mobile PAs require the cooperation of resource users (Hobday & Hartmann, 2006), availability and timely processing of data (Taillon et al., 2012), and freedom for managers to locate reserves as required by the target species (Rayfield et al., 2008).

3.3 Environmental change as an emerging driver of moving targets

With the migration example, species are mobile in space and time, but the movement is often a predictable response to regularly fluctuating resources (Dingle & Drake, 2007). Environmental change, on the other hand, can drive distributional change that is open-ended, on a longer time-scale, and by species that might otherwise be sedentary. These challenges are more widespread and pervasive, and harder to predict.

There are many forms of environmental change that can drive species movements, including habitat modification, fragmentation, and human disturbance. Climate change may emerge as the most consequential of these. The implications of climate change for the future effectiveness of PAs is a topic of enormous research interest (Hannah et al., 2007; Singh & Milner-Gulland, 2011). A changing climate influences not just ecological dynamics, but the human behaviours driving biodiversity loss as well. Climate sets near-absolute bounds upon where species can exist, and interacts with other factors in determining where and at what abundance species actually do exist. Climatic forcing can cause physiological, behavioural, numerical and distributional changes in species, and can do so both directly and through effects on ecological interactions (Parmesan, 2006; Suttle et al., 2007). Additionally, human adaptation in response to climate change may alter our relationship with the surrounding environment in ways that produce further ecological feedbacks (Nicholson et al., 2009). For all these reasons, changing climate regimes will influence the effectiveness of PAs. Poiani et al. (2011) estimate in one study that, of a sample of 20 existing conservation projects, over half would require a change in project focus entirely if climate change impacts were considered.

In cases where species ranges are projected to shift in response to climatic change, mobile PAs could present a solution. For instance, Rayfield et al. (2008) predict that reserves tracking spatial habitat shifts could be effective for the American marten *Martes americana*. In this study, models suggested that such reserves would perform better over 200 years than static reserves. There has additionally been some discussion around designing PA networks that are resilient to climate change through the use of movement corridors and less vulnerable core ‘refugia’ (Malhi et al., 2008), careful spatial planning based on habitat type (Hannah et al., 2007) and projections of habitat suitability (Singh & Milner-Gulland, 2011).

Such approaches are difficult because they require a view into the future. Predictions of ecological responses to climate change involve a number of sources of uncertainty (Walther 2010). These make predictions sufficiently unclear as to demand that conservation takes an adaptive approach (Heller & Zavaleta, 2009). Biodiversity offsets ostensibly demand consideration of ecosystem dynamics, socio-ecological trends and uncertainty, because otherwise NNL cannot be guaranteed. They consequently provide an excellent mechanism through which some of the proposed new approaches to conservation might be implemented.

3.4 Biodiversity offsets and moving targets

Offsets have already been implemented or proposed in ecosystems involving moving conservation targets; though, as I explore in this section, they have not necessarily been designed with this challenge in mind.

3.4.1 *Biodiversity offsets and migratory species*

The appropriate interpretation of NNL is challenging for migratory species. The challenges include whether and how to consider influences on the viability of migratory populations outside of the development and offset areas, and how to account for changes in movement dynamics within the timescale of the project (Table 3.1). These challenges are not always explicitly considered. For instance, Kiesecker et al. (2009) model offsets for the Jonah gas field in the US. The field is underneath desert sagebrush, which provides habitat for the migratory Pronghorn antelope *Antilocarpa americana* and various bird species. The models suggest that most of the objectives of a proposed offset could be achieved over 30 – 50 years using fixed PAs. However the objectives are framed in terms of habitat restoration and protection, under the assumption that transient species will be conserved along with their habitat. As I have discussed, this assumption may not necessarily hold for migratory species that do not spend all their time in PAs and are subject to other threats.

***Table 3.1**: Examples of biodiversity offset schemes that affect migratory species. The table runs roughly in order of increasing consideration given to the mobile nature of migratory species*

'No net loss' target	**Biodiversity Offset objective**	**Example**	**Challenges for mobile/migratory species**
Habitat	Any habitat degraded or lost through development is replaced with created/restored habitat (indirect species conservation is assumed)	*EU 'Natura 2000' sites (McKenney & Kiesecker, 2010)*	Species are not explicitly targeted or conserved, so it cannot be assumed they will be conserved along with their habitat
Habitat used by migratory species	Any area of habitat used by a migratory species that is degraded or lost through development is replaced with created/restored habitat that is also used by that migratory species	*Pronghorn antelope (Kiesecker et al., 2009)*	Habitat type/condition may change with time, e.g. degrade due to climate change. Migratory species may change preference to a different site
Species migration route	Any negative impacts of development upon the migration route of a species are offset by actions that preserve that migration route	*Saiga antelope (UNDP, 2010a)*	Species may change migration route. Species migration might stop entirely
Migratory/mobile species (direct)	Any negative impacts of development upon a population of migratory species are offset by actions that conserve that population	*White-tailed sea eagle (Cole, 2010)*	Species may begin to be impacted by factors that are outside the scope of the offset scheme. The proportion of the population migrating may change
Migratory/mobile species (indirect)	Any negative impacts of development upon a population of migratory species are offset by actions that conserve that species elsewhere in its range/lifecycle	*Seabirds (Wilcox & Donlan, 2007)*	Species may begin to be impacted by factors that are outside the scope of the offset scheme. Difficult to demonstrate equivalence between different stages of a species' lifecycle
Ecosystem function	Any loss of functional value provided by a habitat and associated migratory species following development is restored, via the provision of that or similar habitat/species elsewhere	*US Wetlands (McKenney & Kiesecker, 2010)*	Habitat/species may cease providing function. Habitat/species may provide function somewhere else
Combination of the above	Any losses of habitat, species or ecosystem function following development are compensated for in-kind		Relationship between species/habitat/ecosystem function might change such that offset goals become incompatible, e.g. different species might develop conflicting spatial conservation requirements

Instead of fixed PAs, and using existing offset methodologies, highly mobile species could be targeted wherever in their range they are vulnerable to impacts. Cole (2010) uses a European resource equivalency methodology to design compensation for impacts upon white-tailed sea eagle *Haliaeetus albicilla* populations of the Smøla wind farm, Norway. In this case, the units used are "bird-years" (discounted sea eagle life expectancy). Wind farms cause direct sea eagle mortality in the region. The scheme proposes compensation for new farms by retrofitting existing farms with technologies that greatly reduce sea eagle mortality, resulting in NNL of 'bird-years' relative to the status quo.

Alternatively, offsets could incorporate the approach of targeting the most vulnerable stages in a species' life history, rather than the life stages directly affected by development. For example, Wilcox & Donlan (2007) proposed an offset for seabird bycatch in commercial fisheries by which fishermen would pay a levy for unavoidable seabird mortality, to fund the removal of invasive species from island breeding sites elsewhere in the birds' range. Through bio-economic simulation, Wilcox and Donlan predict that such an approach would effectively achieve NNL.

3.4.2 *Biodiversity offsets and environmental change*

Offsets have arguably been most effective where they have specifically been designed to account for environmental change. This is the case in Victoria, Australia, where legislation requires compensation for cleared native grassland. Native Victorian grassland has been lost from much of its original range, and remnants are now significantly threatened by invasive species that cause deterioration in habitat condition. Offset sites are thus actively managed to suppress or remove invasive species and prevent future decline in condition, as well as restore previously degraded sites. Consequently, because they have been designed with respect to background trends, these offsets could deliver genuine ecological gains in native grassland condition over time within the offset sites (Gordon et al., 2011a). It should be noted, however, that offsets are also generally only effective where practical challenges such as ensuring ongoing monitoring and compliance with the regulations are also being addressed (Bekessy et al., 2010), which may be partly why the Victorian offsets have been relatively successful.

The NNL requirement leads to questions about how the performance of a conservation intervention is defined and measured. Gordon et al. (2011a) find that the perception as to whether NNL has been achieved changes significantly when outcomes are measured against different baselines. For instance, if the offsets are measured against a 'business as usual' baseline of development, grassland conversion and deterioration, then NNL of Victorian grassland is achieved through offsets. However the same offset policy with the same absolute ecological outcomes for grasslands might record losses if compared to a fixed historical baseline, because the managed grasslands are only improving relative to declines elsewhere rather than in absolute terms, conversion is still ongoing outside the offset sites, and the landscape as a whole is deteriorating. The topic of baselines is explored in detail in Chapter 4.

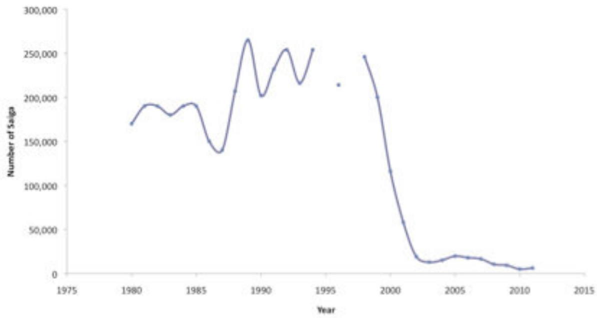
3.5 The saiga antelope in Uzbekistan

Situations in which there is ongoing environmental and social change causing substantial environmental degradation, with additional climate-related change projected, high uncertainty and a paucity of data, and with a threatened migratory species involved, combine to create a particularly difficult conservation problem. Any attempt to define NNL, implement an offset and report against meaningful baselines in such situations could be seen as doomed to failure. Nonetheless an offset scheme is under development in just such circumstances, as mentioned in Chapter 1: a project to compensate for oil and gas development impacts in Uzbekistan, with the migratory saiga antelope as a key conservation target. The example of biodiversity offsets for oil and gas in Uzbekistan provides the primary case study that I use throughout this thesis.

3.5.1 *Background*

The saiga antelope (Fig. 3.1a) has declined by 95% since the fall of the Soviet Union and is listed as Critically Endangered on the IUCN Red List (Fig. 3.1b). The primary driver of this decline is poaching (Kühl et al., 2009; Fig. 3.1c), although environmental change and economic development across its range pose further challenges (CMS, 2010). my focus here is the isolated transboundary population inhabiting the Ustyurt plateau (44°N, 57°E), spanning the border between Uzbekistan and Kazakhstan. The population summers in Kazakhstan, but migrates south into northwest Uzbekistan during winter.

Figure 3.1*. Migration and PAs: (a) saiga antelope (photo N. Singh); (b) the recent population trend in abundance for the Ustyurt saiga population, declining despite having a PA in the Uzbek portion of its range; (c) a saiga that was poached for horn in its Uzbek winter range, in an area without effective protection*

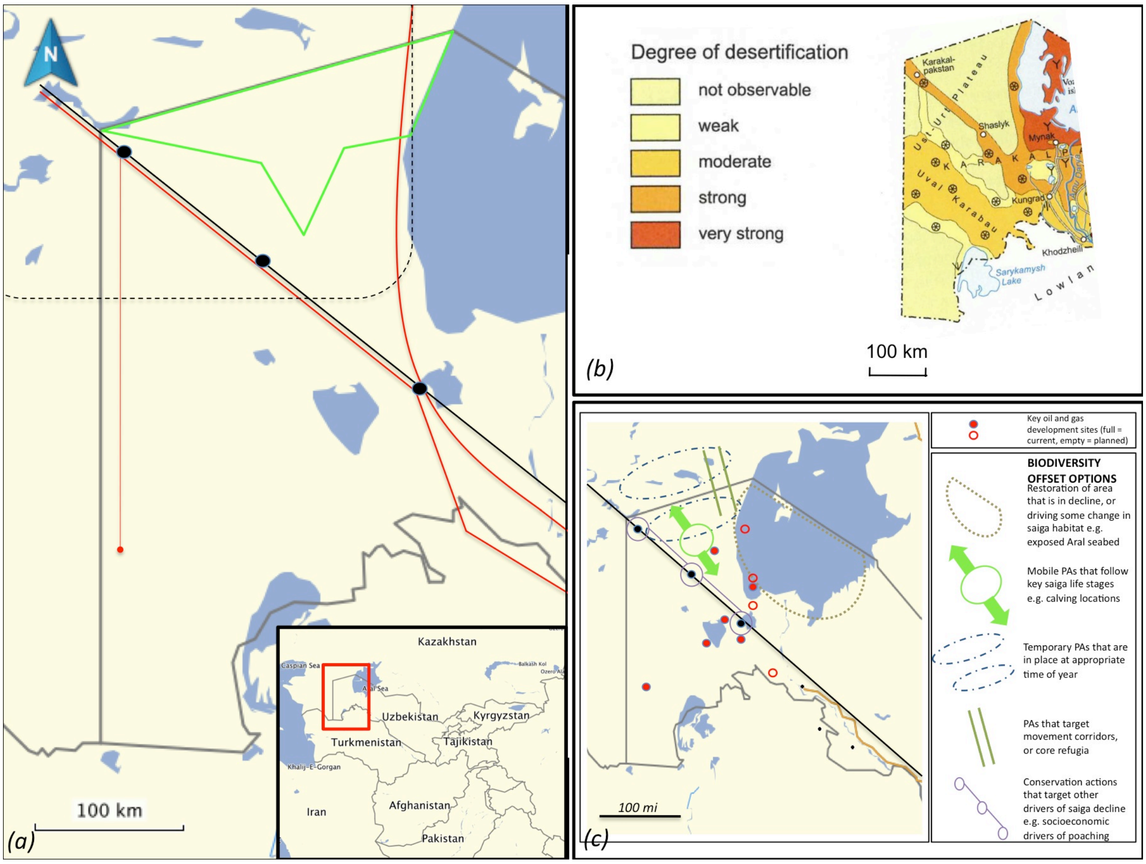


Saiga migration is driven by climate and forage availability (Singh et al., 2010a). Saigas have shifted their calving sites northwards in recent decades; in response to changes in climate and forage (Singh et al., 2010b). The Intergovernmental Panel on Climate Change (IPCC) project that temperatures will rise significantly over the coming century, by as much as 6^0C in Central Asia (Fig. 3.2a), which would very likely force further regional shifts in saiga habitat use. Considering these points, the combined impact of climate change and poaching upon the distribution and migration patterns of the Ustyurt saiga may prove significant (Singh & Milner-Gulland 2011).

Figure 3.2. *Environmental change: (a) sun over the Uzbek Ustyurt plateau, late winter. Winter temperatures have trended upwards over the last 30 years; (b) Alhaji spp. (right), an excellent livestock forage crop, alongside Piganum spp. (left), a poor forage crop and indicator of rangeland degradation*

The Ustyurt has experienced major environmental change in recent history. It is bordered to the east by the Aral Sea, which has halved in area since the 1950s, as a direct result of irrigation (Fig. 3.3b). This has created more than 40,000 km^2 of polluted and saline desert by exposing the seabed. The prevailing wind has deposited significant amounts of dust from the seabed upon the plateau (Ataniyazoza, 2003). Since the collapse of the Soviet Union, livestock practices have changed and they are now at high density near human infrastructure but relatively absent from more distant pastures. This process of grazing redistribution is leading to shifts in plant species composition (Opp, 2005, Fig. 3.2b). These processes are likely to continue influencing landscape condition in the future, feeding back into the effectiveness of any regional conservation and sustainable development schemes.

Figure 3.3: *(a) Schematic of case study area of interest, the Ustyurt plateau in northwest Uzbekistan. Red lines = oil and gas pipelines; black line = railway; dashed area = saiga range in winter; green area = "Saigachy" reserve. Larger map inset. (b) desertification in the region, modified from Opp (2005). Note in red the previous extent of Aral Sea as shown in (a, c), which is now exposed seabed. (c) schematic of the area of interest, with some of the dynamic conservation actions suggested in the text.*



The capacity of saigas to respond to this environmental forcing is constrained by growing anthropogenic disturbances on the Ustyurt (Fig. 3.3a). Saigas react strongly to disturbance, and this will increase with expanding natural gas development, and infrastructural intensification on the Kazakh side of the border including new roads and border fencing. Uzbekistan is in the world's top 20 natural gas producing nations (Effimoff, 2000), containing 194 confirmed oil and gas reserves with a combined economic potential of $1tn USD (UNDP, 2010a). The Ustyurt and Aral region forms one of two main gas extraction regions in the country (EIA, 2012). There are currently at least 5 foreign companies in product sharing agreements with Uzbekneftegaz, the state gas company, including more than 10 fields and $3bn USD of foreign investment for gas extraction and exploration activities in the Ustyurt and Aral Sea region (EIA, 2012; UNDP, 2010a).

3.5.2 *Making offsets a dynamic conservation tool*

The Uzbek government is working with the United Nations Development Programme to investigate mitigation and compensation measures for reducing the biodiversity impacts of the extractive sector, including offsets. The broad objective of offsets would be to create conservation zones with ecological values equal to or exceeding those lost due to industrial development. One proposal is for companies active in the area to fund 80% of the costs of management and anti-poaching enforcement in a re-designated *Saigachy* PA, as compensation for direct impacts elsewhere upon the Ustyurt (UNDP, 2010a; Fig. 3.3a). Saigachy is at present in a slightly different location to that proposed (Esipov et al., 2009), and effectively inoperative.

The Saigachy PA would protect the semi-arid desert habitat of the plateau and associated species assemblage, with saigas as the flagship conservation target for the region; companies would provide equipment and financing for the reserve for the lifetime of their development activities. Without the resources provided to enforce the reserve, there will be no effective protection, and the saigas are likely to continue to be hunted illegally throughout their winter range in Uzbekistan. Therefore, whilst the offset would not create additional biodiversity, it would prevent impacts from multiple sources unrelated to oil and gas activity, thereby avoiding potentially significant projected losses.

So, one key objective for the proposed offset is to prevent further decline and potential extirpation of the saiga in Uzbekistan. The NNL requirement means that current and future abundance and population dynamics of the species must be assessed. This in turn makes clear that simply protecting the saiga in a fixed PA in Uzbekistan is insufficient as the saigas are also heavily poached in Kazakhstan (Kühl et al., 2009). Consequently, as offset actions are not limited to fixed PAs, the offset should incorporate other conservation activities. It is also clear that the problem of conserving the saiga along with their habitat must be viewed in a landscape context, rather than simply at development project level.

One option would be to fund anti-poaching measures in Uzbekistan during the winter and in Kazakhstan during the summer. Another might be to fund mobile PAs that track the most vulnerable parts of the saiga lifecycle i.e. calving locations. Studies addressing the spatio-temporal dynamics of poaching will help to identify areas in which to enforce protection or place PAs. If PAs must be fixed, then an ecological network approach might be considered, for instance focusing on movement corridors and refugia. Alternatively, as discussed by Cole (2010), conservation effort could be targeted at the points of species impact: e.g. further measures could be taken to prevent poachers from using known trading routes for saiga products (such as the railway) or to discourage inhabitants of the few settlements that exist in the Ustyurt from poaching. This might involve all manner of interventions, from training enforcement dogs to locate saiga horn, to creating alternative livelihoods for poor families that might otherwise poach. Offsets involving an internationally integrated approach like this could be possible within the framework of the international Memorandum of Understanding on saiga conservation, signed by both Kazakhstan and Uzbekistan under the Convention on Migratory Species (CMS, 2010).

It may be that saigas are not an adequate proxy for the condition of biodiversity on the Ustyurt plateau and that instead, for instance, offsets should focus on habitat conservation. In this case, achieving NNL would again require consideration of regional environmental change. The gradual deterioration of the habitat due to salinisation and poor grazing regimes provides a baseline for NNL, in which case, implementing measures to better manage grazing might be appropriate. On a larger scale, funding directed towards reducing or reversing the loss of the Aral Sea itself, as explored in Kazakhstan through improved water and irrigation management, could have positive conservation outcomes for the Ustyurt. Of course, given gas exploration opportunities under the exposed Aral seabed, this might be unlikely to receive support from those funding the offsets. The dynamic conservation options discussed in this and the previous paragraph are displayed schematically in Fig. 3.3c.

Whilst these and other measures (Table 3.2) are in theory feasible unless stated, considerable work will be required to realize them. In particular, national legislation in Uzbekistan does not currently contain the measures needed to implement offsets successfully; there is a need to amend the laws on Subsurface Resources, Territorial Planning, and Protection and Use of Flora and Fauna to include a mitigation hierarchy and biodiversity assessment guidelines (UNDP, 2010a). Further, there have clearly been difficulties in ensuring compliance with anti-poaching legislation in the past in Uzbekistan and Kazakhstan, and this will continue to be problematic unless it is directly addressed, either within the offset or independently.

***Table 3.2**: Some problems and potential solutions for the conservation of moving targets, illustrated using the Uzbek biodiversity offset as a case study.*

Biodiversity offset objective	**General problem with moving targets**	**Specific Uzbek case study challenge**	**Possible solutions**	**Comparative examples**
Species (*Saiga tatarica*): saigas do not decrease in population abundance or spatial distribution	Species significantly impacted outside of offset landscape/region	Poaching continues outside of Uzbekistan (in Kazakhstan)	Implement standard fixed PA offset, monitor direct reductions in saiga poaching within PA	*Thirgood et al., 2004 Kiesecker et al., 2009*
			Coordinate PA offsets across entire species range (i.e. including Kazakhstan)	*Apostalaki et al., 2002 Martin et al., 2007*
	PA offset does not target sensitive part of species' life history	Calving sites located outside of PA offset (in Kazakhstan)	Coordinate PA offsets across entire range (i.e. including Kazakhstan)	*Apostalaki et al., 2002 Martin et al., 2007*
			Implement mobile PA offsets that track calving locations	*Robinson et al., 2008*
	Species migration shifts out of PA offset area, or stops entirely	Anthropogenic activity/climate drives changes to saiga migration pattern and habitat use	Implement temporary PA offsets that track saiga migration corridor	*Rayfield et al., 2008 Shillinger et al., 2008*
	Species move outside of PA, and are negatively impacted by people/industry	Continued poaching in Uzbekistan	Offsets provide resources to reduce demand for saiga horn/meat, or reduce poaching through alternative livelihoods and public engagement	*Kühl et al., 2009*
		Direct mortality from industrial or infrastructural expansion	Offsets provide resources to prevent or mitigate direct impacts on species	*Cole, 2010*
Habitat type (Ustyurt plateau): habitat is not significantly reduced in area, habitat condition, or functional condition	Habitat degraded by change in inorganic or organic composition,	Ustyurt undergoes increased salinisation, desertification	PA offset objectives defined against dynamic baseline of increasing salinisation, sites protected against further losses	*Gordon et al., 2011a*
			PA offset objectives defined against fixed baseline, and include active management (e.g. vegetation restoration, grazing management)	
	Vegetation structure changes	Predicted temperature rises cause change in growing season/bioclimatic niches	PA offset objectives defined against regional baseline that is mobile due to climate change	*Gordon et al., 2011a*
			PA offset objectives defined against fixed baseline, and PA incorporates measures to be resilient to climate change	*Malhi et al., 2008 Hannah et al., 2007*
	Habitat degraded via direct anthropogenic impacts	Stocking regimes (over or undergrazing) detrimental to rangeland condition	PA offset is designed with adaptive management of feedbacks between social and ecological systems (e.g. livestock grazing regimes)	

3.5.3 *Summary*

This case study illustrates the challenges I have discussed concerning conservation in a dynamic world. The currently proposed offset scheme would primarily result in a single fixed PA that does not cover all parts of the target species range and lifecycle crucial to preventing further declines. As saigas are migratory, and the plateau is changing, this exemplifies the application of a static conservation tool to dynamic targets. However, in explicitly considering the mobile nature of both species and habitat, proposals for using an offset framework to test more dynamic conservation approaches could be explored (Table 3.2). The necessary institutional frameworks and understanding may exist to some extent, as discussed further in Chapter 5. More detailed research and adaptive management would be necessary to establish which of the dynamic approaches suggested here and in Table 3.2 would be effective, especially given practical considerations such as ensuring compliance. Equally, political considerations would inform discussion around using trans-boundary offsets. However, the offset scheme offers a framework into which these more dynamic approaches can be incorporated, explored and implemented.

3.6 Conclusions

Fixed PAs - which exemplify the still largely static approach to conservation - are insufficient to protect biodiversity as the world changes increasingly rapidly. We must find dynamic new ways to conserve nature.

There is a growing theoretical basis for ideas such as mobile PAs, management of habitat to prevent declining condition against future baselines, and conservation efforts that address vulnerable parts of species' lifecycles and the shifting incentives that resource users face. These dynamic approaches may prove considerably more effective than traditional static conservation. Now we must explore them through practical implementation, rather than just through the thought experiments and simulations which have dominated up to now.

Biodiversity offset schemes are ostensibly structured in a way that can address the dynamic nature of conservation targets. By requiring NNL of biodiversity over a specified time period, offset schemes lend themselves towards taking spatial and temporal dynamics into account. Because of this, and due to their widespread take-up, offsets now present an opportunity to implement and test dynamic conservation approaches in the field, as I illustrate in the case study. These new approaches can be designed into offset schemes in an adaptive manner, with evaluation of their effectiveness as an integral component of the offset itself. This will require the strong commitment of all concerned to true compliance with the underlying rationale of offsets, rather than paying lip service to the concept of NNL. But by exploiting this opportunity to learn, we may begin to be able to conserve moving targets – a crucial aim in a changing world.

Chapter 4

The importance of the frame of reference in evaluating conservation interventions and achieving no net loss of biodiversity

“Anyone with gumption and a sharp mind will take the measure of two things: what's said and what's done.”

Anonymous (from “Beowulf”, 8th – 11th Century AD)

“Yea, foolish mortals, Noah’s flood is not yet subsided; two thirds of the fair world it yet covers.”

Herman Melville (from “Moby Dick”, 1851)

4.1 Introduction

When setting objectives for any conservation activities, or judging their efficacy after implementation, an appropriate frame of reference against which evaluation is made should be specified. This would mean considering the eventual state of the region in which conservation activities have taken place and comparing it with some known historical state (a baseline) and against some alternative scenario that would have taken place without the intervention (a counterfactual). That appropriate frames of reference are not widely used in practice is a problem for contemporary conservation (Ferraro & Pattanayak, 2006; McDonald-Madden et al., 2009). Often, if a region is subject to some conservation intervention, then subsequent ecological recovery is treated as a success and further deterioration as failure. Consequently, interventions are implicitly evaluated against a baseline consisting of the fixed point in time at which intervention began, if at all. Yet a fixed baseline may be an insufficient basis for judging the true impact of interventions because ecosystems are dynamic (Nicholson et al., 2009) and can change without intervention (Ferraro & Pattanayak, 2006).

The actual success or failure of an intervention depends on the frame of reference chosen to assess it against. For instance, if a region's biodiversity is decreasing in some way that conservation seeks to address, then slowing that deterioration may represent a positive outcome, even if deterioration continues in absolute terms. Likewise, conservation activity imposed on a region that is recovering from some decrease in biodiversity may be misjudged as having succeeded, even if the overall trend for recovery was not actually altered. For example, tropical forests are protected to prevent deforestation, and an expansion in protected area may be considered a success for conservation. But their effectiveness depends on how much forest would have disappeared in the absence of protection (i.e., the counterfactual scenario), which is often ignored (McDonald-Madden et al., 2009). Protected areas are instead often evaluated based on the amount of forest left standing (Andam et al., 2008). Performance evaluation of conservation interventions should involve clearly specified counterfactuals that incorporate consideration of the likely trajectory of the target region without the intervention.

In addition to evaluation of outcomes, the choice of reference frame can shape how an intervention is structured and implemented. For example, 74% of the Aral Sea by area has changed from saline lake to semi-arid scrubland over recent decades (Micklin, 2007). In this case, the difference between a 1960 and a 2000 baseline as a frame of reference for conservation activities is important. So the choice of reference frame is a critical component of the process of conservation, even if it is often unstated and implicit only in conservation policy.

Reference frames are commonly applied in policy relating to climate change. Under the Kyoto Protocol, participating countries agreed to reductions in the annual percentage of greenhouse gas emissions relative to 1990 levels (UN, 1998). Thus, emissions of greenhouse gases in the year 1990 are a baseline against which change is evaluated. A counterfactual could also be used; for example, actual annual emissions could be compared with calculated annual business-as-usual emissions (i.e. had the protocol never been implemented and the status quo maintained). Climate change policy also highlights the difficulties in specifying reference frames. For instance, the baseline could be adjusted to account for industrialization, incorporating a development adjustment factor to allow higher emissions for poorer countries so as not to limit socio-economic development (Angelson, 2008). The inclusion of such a factor would place a greater burden on industrialized nations to reduce emissions and increase the effort required to successfully implement climate policy. Clearly, choosing between reference frames may be controversial.

In conservation, the relationship between reference frames and outcomes has been implicitly discussed since the introduction of the concept of shifting baseline syndrome in the 1990s (Pauly, 1995). The conjecture is that each successive generation of conservationists sets extant biodiversity during their early years as their own personal baseline for what is natural; thus, they mask longer-term biodiversity decline. The topic of appropriate baselines also arises in restoration and rewilding literature (Alagona et al., 2012), but altogether it receives less attention in conservation than in climate policy. Even in newly developing conservation approaches, such as biodiversity offsetting, baselines can be overlooked (Quétier & Lavorel, 2012; Chapter 2). Given a specified frame of reference, however, scientists can tell us whether achieving a stated conservation objective is feasible and what might influence outcomes, as explored here.

4.1.1 *Defining a frame of reference*

I use the term reference frame to represent a state against which conservation interventions can be evaluated through some measure of biodiversity, of which baselines and counterfactuals are both types. The choice of reference frame is ultimately a value judgment, yet there are criteria that should guide reference frame specification and practical interpretation. A useful reference frame must include at least two facets of environmental change in the focal region: ongoing trends in biodiversity (i.e., biodiversity trajectory) and human impacts upon biodiversity (i.e., anthropogenic impacts) (Table 4.1).

Table 4.1: *A description of the contexts for biodiversity-loss offset interventions in a dynamic socio-ecological system.*

Context	Applies to	Example options	Explanation
Anthropogenic impacts + conservation intervention	anthropogenic pressure upon target ecosystem	no development	no development, no offsets
		development only	development has negative impacts upon biodiversity, but no offsets implemented
		development with offsets	development has negative impacts upon biodiversity, offsets implemented to compensate
Biodiversity trajectory	likely biodiversity trend in the absence of anthropogenic pressure	decreasing	biodiversity would deteriorate with time, e.g. invasive weeds displacing existing species
		stable	biodiversity would remain stable (e.g. a climax habitat type)
		increasing	biodiversity would improve with time (e.g. previously exploited species increasing in abundance)

The biodiversity trajectory is the trend in biodiversity within some defined region in the hypothetical absence of any further anthropogenic development or conservation activity (including drivers caused by past anthropogenic activities or processes, such as climate change or invasive species). For instance, remnant native grasslands in Victoria, Australia, would continue to decline in conservation value in the absence of subsequent human activity, driven partly by generalist invasive species. Therefore protection from urban development – a current anthropogenic threat – is an inadequate conservation strategy for these remnants; they must be actively managed for conservation of native species (Gordon et al., 2011a).

Anthropogenic impacts are the effects of human activities taking place within the region that affect biodiversity, positively or negatively. These include negative impacts upon biodiversity from development and land use change and existing projects to safeguard biodiversity that are not part of the conservation intervention to be evaluated.

The biodiversity trajectory and existing anthropogenic impacts provide necessary background for new conservation interventions. They determine whether a baseline or counterfactual is the most appropriate frame of reference and are used to develop robust counterfactuals. Three other elements of a frame of reference must be specified: spatial scale, temporal scale, and time lags between losses and gains in biodiversity.

Frames of reference can be set at the scale of actual intervention projects (e.g. patches of vegetation directly manipulated) or of the larger landscape (e.g. manipulated patches plus surrounding areas in which interventions are not directly undertaken). Either scale may be reasonable dependent upon the situation. For example, EU agri-environment schemes (Whittingham, 2007) can be evaluated by aggregating information from only individual farms involved or from the entire landscape, including areas not part of the scheme. The evaluation of the schemes success may be scale dependent: biodiversity may increase on individual farms (i.e. at the project scale), whereas regional (i.e. landscape scale) changes in biodiversity may be negligible (Whittingham, 2007).

The reference frame can be fixed at some baseline point of initial measurement, such as the outset of the intervention, or based on predicted counterfactual trends through time. These can be thought of as fixed and relative frames of reference, respectively. The biodiversity baseline in the region at the time the intervention began is often the frame of reference specified or implied for conservation initiatives. Conversely, counterfactuals may include the biodiversity trajectory for the region had there been no development or intervention (i.e. the status quo or no-development scenario) or the worst-case scenario of development without compensatory conservation (i.e. development only). Whilst both types of reference frame are subject to uncertainties, counterfactuals are inherently more uncertain because they involve predicting future trends in addition to taking measurements (Ferraro & Pattanayak, 2006; Pouzols et al., 2012).

Time lags are important in conservation (Maron et al., 2012). Development may result in immediate biodiversity losses, but ecological gains from compensatory conservation activities may take time to accrue. Time lags are undesirable if the existence of biodiversity provides some ongoing ecosystem service, which is diminished during the time lag (Bekessy et al., 2010). Then, even if biodiversity levels were eventually restored to pre-development levels, the ecosystem services lost in the interim could necessitate additional compensation. Time lags may be more or less important depending upon when the intervention is evaluated (e.g., whether it is assessed 1 year, 10 years, or 100 years after the initial intervention). Although not part of the frame of reference per se, the point in time at which interventions are evaluated is critical because this often has implications for the evaluation outcome.

4.1.2 *Exploring different frames of reference for biodiversity offsets*

I use biodiversity offsets as an example through which to explore the effects of reference frame specification on conservation outcomes. Again, biodiversity offsets (‘offsets’) are interventions that: provide additional substitution or replacement for unavoidable, negative impacts of human activity on biodiversity; involve measurable, comparable biodiversity losses and gains; and demonstrably achieve, as a minimum, no net loss of biodiversity (Chapter 2). Offsets are useful for an exploration of reference frames because they have clearly articulated objectives: no net loss (NNL) or a net gain (NG) in biodiversity (McKenney & Kiesecker, 2010; Madsen et al., 2011). To demonstrate NNL or NG fundamentally requires definition of a frame of reference against which to evaluate losses and gains (Gordon et al., 2011a).

The overriding objective was to explore key considerations when specifying reference frames and to determine how these affect the outcomes of conservation interventions, through the lens of biodiversity offset models. I considered an intervention successful when an offset intervention secured NNL of biodiversity. This can be measured against various reference frames, based on different biodiversity trajectories and anthropogenic impacts. I examined the implications of choosing different reference frames for interventions. I defined reference frames at the project and landscape scales and specified them as a baseline or counterfactual. I considered the impact of time lags and how the difficulty in achieving conservation success changes with different reference frames. I used two types of model. The first was a general (aspatial) model that allowed examination of the best-case performance of a generalized offset policy against a set of reference frames combining three different biodiversity trajectories and three different anthropogenic impact scenarios. I also considered uncertainty in this model by assuming incomplete knowledge about the parameters governing the biodiversity trajectory. I used the second (spatial) model to test the conclusions for the combination of trajectories and impacts associated with a real example: urban development and offsetting within deteriorating grassland ecosystems around Melbourne, Australia (Gordon et al., 2011a).

4.2 Methods

4.2.1 *General model*

I created a generalized biodiversity offset model to explore the relationship between biodiversity trends and anthropogenic impacts when the relationship is evaluated against different frames of reference (Tables 4.1, 4.2). The model is based upon a function B, representing the biodiversity of some region, where B is a function of time t, and at $t = 0$, $B(t) = B_0$, a constant representing initial biodiversity value. The absolute values of the model parameters are arbitrary, and those used to generate my results are in the Appendices (A1.1). The period for evaluating offset outcomes was 100 years, and the parameters were set so that all biodiversity in the region was developed or offset within this period.

***Table 4.2**: Key spatial and temporal factors that should be considered when specifying a frame of reference for measuring the performance of a biodiversity-loss offset intervention*

Consideration	Applies to	Example options	Explanation
Spatial scale	total area to include within the NNL calculation	Project	biodiversity losses & gains compared across development & offset project sites only
		landscape	biodiversity losses & gains summed across the entire region, i.e. including the matrix of sites which are neither development nor offset
Temporal scale	offset success assessed against a current or projected level of biodiversity	fixed (baseline)	biodiversity value at some point in time measured or estimated and considered the baseline; NNL is assessed against this fixed baseline
		relative (counterfactual)	NNL is assessed against predicted future trend in biodiversity
Time lag	temporary losses in biodiversity value, between development occurring and offset maturing	include interim loss in biodiversity in baseline; do not include interim loss in biodiversity	possible to either include or exclude the summed biodiversity benefit lost due to time lags from calculations as to whether conservation objectives have been achieved. This is a consideration not explicitly included in the generalised model developed here, but time lags are evident in the Melbourne case.

The quantity B(t) is determined by three functions: dev(t), the amount of biodiversity lost to development over time; off(t), the gain in biodiversity from offsets over time (in response to development); and T(t), which describes the underlying biodiversity trajectory. In this model, the anthropogenic impacts are represented by dev(t) and the biodiversity trajectory by T(t). In the absence of development and offsetting the biodiversity trajectory is given by

$$B(t) = T(t) \times B_0. \tag{4.1}$$

With both, the biodiversity trajectory can be written as

$$B(t) = T(t) \times [B_0 - dev(t)] + [p(t) \times off(t)]. \tag{4.2}$$

The function p(t) specifies how the (protected) biodiversity contained in offset locations changes over time in response to offset actions. For simplicity, I assumed the managed biodiversity within the offset remained constant (p(t) = 1). A more general application of this model could explore other functional forms for p(t), or couple p(t) with T(t).

In the absence of any intervention, I assume biodiversity in the region can follow one of three trajectories: decreasing, stable, or increasing over time. The stable trajectory was modeled by a constant and the decreasing and increasing trajectories as logistic curves based upon the functional form described in Mace et al. (2008) for population decline:

$$T(t) = 0.5 + \frac{1}{(1 + e^{k_1 \times t})} \tag{4.3}$$

The coefficient k_1 determines whether the trajectory is decreasing (positive k_1) or increasing (negative k_1) and its value determines the shape of the logistic function (i.e. how quickly the biodiversity component decreases or increases). These functions provide an approximate representation of biodiversity change (Mace et al., 2008). Results with other functional forms are in Appendix 1 (A1.3).

We assumed a linear loss of biodiversity from development over time, occurring at a constant rate determined by the parameter k_2, which was negative

$$dev(t) = k_2 \times t. \tag{4.4}$$

Different types of development could be modeled by substituting different functional forms into Eq. 4.4. Offsets associated with development were expressed as

$$off(t) = -m \times dev(t) = -m \times k_2 \times t. \tag{4.5}$$

Because development impacts dev(t) are negative, I included a factor -1 in Eq. 5 so that offsets represented a positive gain for biodiversity. I made the optimistic assumption that offsets occur simultaneously with development and create new biodiversity immediately. There was no limit to the amount of biodiversity that could be added to the region, so development impacts could always be offset. The factor m in Eq. 4.5 multiplies the amount of biodiversity offset for a given development, meaning that if m = 2, twice the biodiversity lost from development would be created by the offset. Values of m < 1 represented the case in which offsets were only partially successful and thus created less biodiversity then development removes. Unless specified otherwise, m = 1 for all simulations. I also assumed that, once created, offsets are managed in perpetuity and remain of constant biodiversity no matter what form T(t) takes.

The parameters B_0, k_1, k_2, and m and the functions p(t), T(t), dev(t), and off(t) will all have uncertainties associated with them in a real ecosystem. I simulated the effect of uncertainty by varying the values of k_1 and m; a more thorough exploration of uncertainty is beyond the scope of this study.

I generated three different anthropogenic impact and intervention scenarios based upon the above equations for comparison of offset performance: no development (dev(t) = off(t) = 0, so $B(t) = T(t) \times B_0$); development only (development occurs without offsetting [m = 0], so $B(t) = T(t) \times [B_0 - dev(t)]$); and development with offsets (development and offsets occur, so B(t) is given by Eq. 4.2).

In combining these three anthropogenic impact scenarios with the three different biodiversity trajectories, hypothetical systems subject to different dynamics were created. I examined how B(t) changed over time for each anthropogenic impact scenario and biodiversity trajectory. Further details on assumptions are in Appendices (A1.1). Although the three scenarios above represent the approach taken to regional biodiversity management, they can also be used as counterfactual frames of reference.

4.2.2 *Real world simulation model*

The general model presented above depicts an idealistic offset process with several simplifying assumptions. To translate this into the real world, I utilized an existing model developed for biodiversity offsets associated with the clearing of native grassland to expand the city of Melbourne (Gordon et al., 2011a). In the Melbourne model, the following simplifying assumptions are dropped: that offset gains are instantaneous and simultaneous with development, offset areas remain constant in biodiversity value, and biodiversity value can be created without limit in a landscape. Consequently, managed offset areas gradually improved rather than remaining stable, but there was a limit on the total amount by which B(t) could be increased.

The spatially explicit model was coded in R (R Development Core Team, 2012) and developed for a region west of Melbourne. It begins with a map depicting the condition of native grassland, derived from satellite data, upon which cadastral land parcels are overlaid. The summed condition of native grassland across all parcels in the landscape is a real world equivalent of the value B(t) used in the general model. To model anthropogenic impacts and conservation interventions, land parcels are sequentially developed and then offsets are implemented to compensate for the resulting loss of grassland biodiversity. Offset criteria are based on rules derived from state of Victoria legislation, and require that the area multiplied by the condition of the grassland developed must result in an offset m times larger, such that:

$$(\text{area x condition})_{\text{offset}} \geq \text{m x } (\text{area x condition})_{\text{developed}} \tag{4.6}$$

In the model of biodiversity trajectory, at each time step, the condition of native grassland evolved on different trajectories depending on current condition of the grassland and whether or not it was being managed as an offset (Appendix A1.2). Stochastic variation was incorporated into the model by including small random fluctuations in condition B(t) at each time step and by selecting parcels for offset and development randomly (but subject to constraints; see Appendix A1.2 for details). I ran the model 25 times under each scenario and examined the average behavior of biodiversity condition B(t) over time. Under the Victorian scheme, 'habitat hectares' (HH) is used as a metric for measuring habitat condition (Parkes et al., 2003). The quantity B(t) is therefore measured in HH condition scores summed across all land parcels in the region. But because my results are not specific to a particular biodiversity metric, I do not report HH values.

I extended the original Melbourne model in two ways. First, the biodiversity trajectory of unmanaged grassland was varied. In this model, a grassland biodiversity deterioration curve was derived from expert opinion and applied to all parcels of unmanaged land. I also explored the consequences of assuming the biodiversity trajectory was stable. The stable trajectory in the Melbourne case was unrealistic, but I used it to corroborate the outcomes of the general model. This was justifiable because my primary aim was not to provide a realistic assessment of the outcome of offset policies for Melbourne, but to explore the implications of the choice of reference frame. Second, the output was examined against the different frames of reference in Table 4.2 to determine how the choice of reference frame affected offset performance.

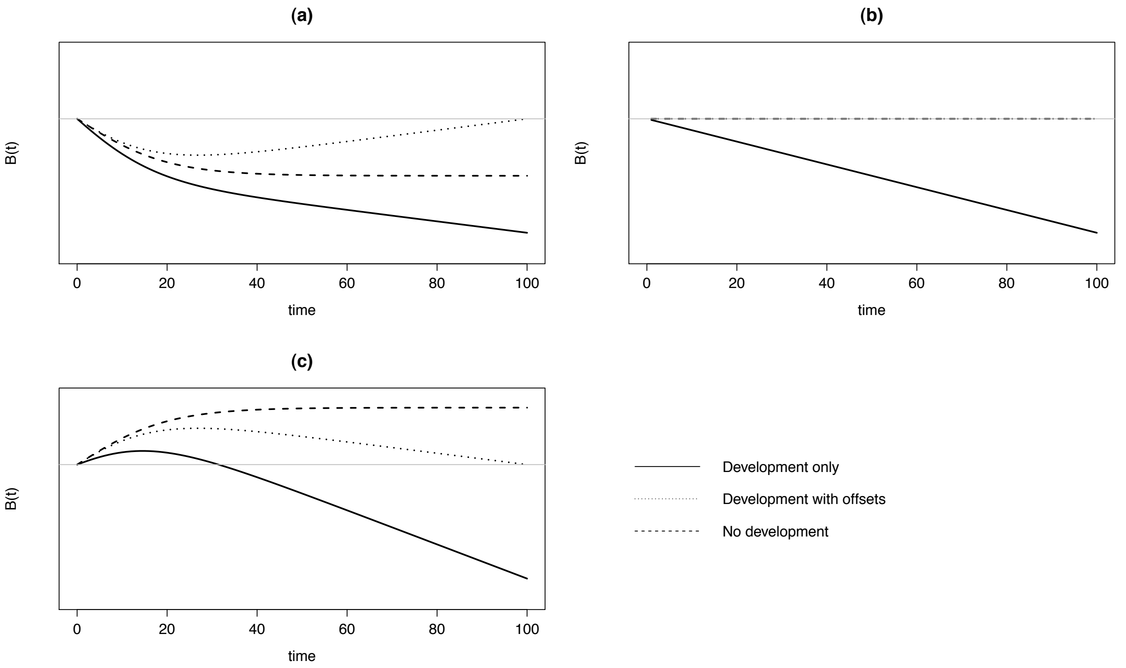
To explore the importance of the spatial scale of the frame of reference, I noted that summing biodiversity value across all land parcels reflected evaluation against a landscape-scale frame of reference. I then assessed outcomes at a project scale, done by summing B(t) across only those parcels that were either offset or developed.

4.3 Results

4.3.1 *General model*

In all cases, against a fixed baseline, biodiversity offsetting eventually resulted in no net loss of biodiversity (NNL). However, the conservation outcomes in the interim varied significantly depending on the trajectory of the ecosystem and led variously to a net loss (NL), NNL, or a net gain (NG) depending on whether the biodiversity trajectory was decreasing, stable or increasing, respectively (Fig. 4.1). Therefore, despite that NNL was achieved after 100 years for all three trajectories, if the offset policy were evaluated or abandoned after 50 years, the outcomes achieved would change depending upon the biodiversity trajectory.

Figure 4.1*: (a) Decreasing, (b) stable, and (c) increasing biodiversity (B[t]) over time under the three development scenarios in a general theoretical model of a hypothetical ecosystem (horizontal line through the origin represents the fixed baseline). Results were calculated at a landscape scale. No metric is specified here for biodiversity, so no scale is given, but biodiversity increases above the origin.*



Under a decreasing biodiversity trajectory, the development-with-offsets scenario outperforms the no-development scenario, whilst this is reversed under an increasing trajectory (Table 4.1; Fig. 4.1a, 4.1c). This was a result of evaluating against a fixed baseline. When biodiversity was decreasing, development with associated offsets initially only slowed down the decline (Fig. 4.1a). When biodiversity was already increasing, development with offsets only hampered improvement due to the loss associated with development impacts (Fig. 4.1c). With the stable biodiversity trajectory (Fig. 4.1b), development with offsets and no development scenarios had identical performance due to my assumption that each offset created new biodiversity equal to that what was lost to development.

With an NNL objective, the choice of reference frame completely determined whether and when the offset intervention was successful. In a deteriorating ecosystem (Fig. 4.2), offsets eventually lead to NNL against the no-development or development-only counterfactuals, over the period modeled (Fig. 4.2b, 4.2c). Against the fixed baseline, the development-with-offsets scenario also eventually produced NNL and outperformed the no-development counterfactual (Fig. 4.2a). However the fact that I assumed biodiversity remained stable rather than increased within the offset area meant that overall, biodiversity still decreased for the majority of the simulation (Fig. 4.1; Table 4.3).

***Figure 4.2**: Outcomes for general model of a hypothetical ecosystem showing biodiversity (B[t]) that is on a decreasing background trajectory, measured against different frames of reference as follows: (a) fixed baseline, (b) no-development counterfactual, (c) development-only counterfactual. The development-with-offsets intervention is considered a net loss of biodiversity until the 100-year mark with a fixed baseline, it is considered a net gain in biodiversity at all times for both counterfactuals (Table 4.3).*

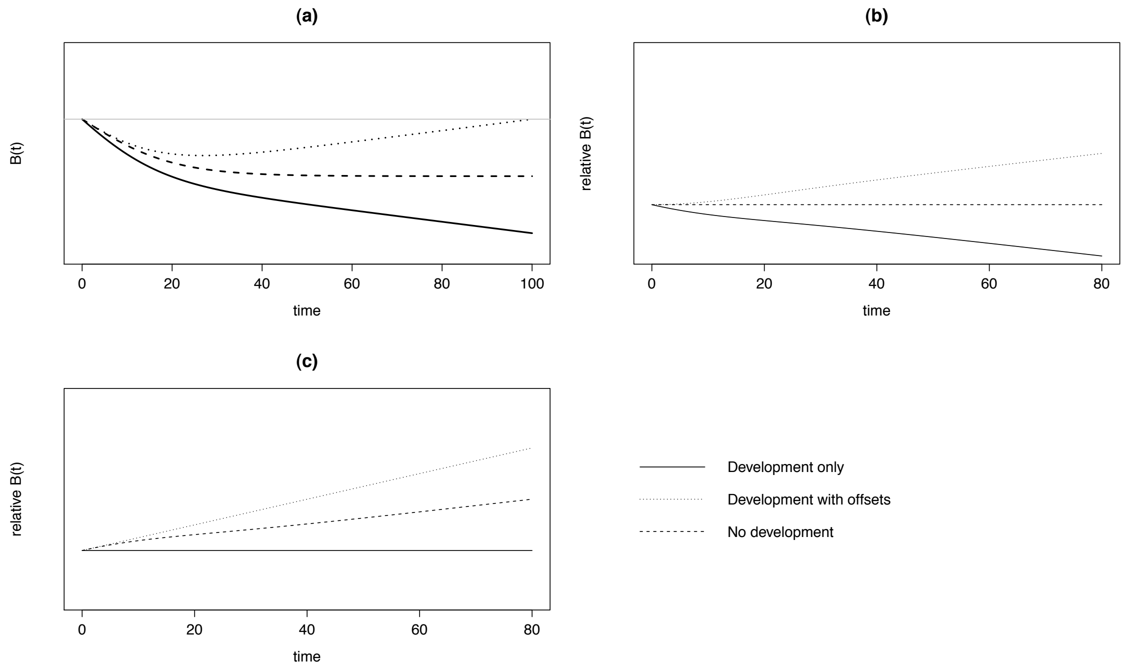


Table 4.3: *The best possible outcomes for biodiversity-loss offset schemes under three different biodiversity trajectories, against both fixed and relative frames of reference.**

Biodiversity trajectory	Fixed current baseline	No development counterfactual	Development -only counterfactual
Decreasing	–	+	+
Stable	0	0	+
Increasing	+	–	+

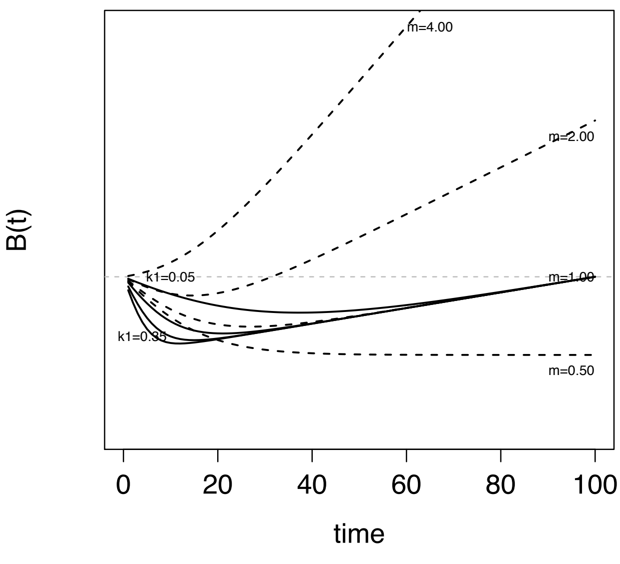
** Results calculated at the landscape scale. Shown is overall biodiversity change within the first 100 years for the development plus offset scenario relative to each frame of reference. Key: –, net loss; 0, no net loss; +, net gain.*

Performance against a counterfactual varied, depending on which alternative scenario was defined as the relative frame of reference. Against a no-development counterfactual, offsets improved the situation in deteriorating regions and made it worse in improving regions (Table 4.3). Against a development-only counterfactual, offsets always led to a NG, regardless of the underlying biodiversity trajectory (Fig. 4.2; Table 4.3).

The results presented here were obtained assuming logistic functional forms for biodiversity trajectory. When repeated with a range of alternative forms, the results were qualitatively the same, as reported in Appendix 1 (A1.3).

With decreasing biodiversity and a fixed baseline, varying k_1 (Eq. 4.3) resulted in initial variation in conservation outcomes that then converged over time (Fig. 4.3). Conversely, varying m led to conservation outcomes diverging over time. This result was partly due to my implicit assumption that all biodiversity is eventually managed via offsets or lost through development, so over longer time scales the outcomes were more sensitive to offset multipliers than biodiversity trajectory. However, this may be realistic for some landscapes subject to continued development.

Figure 4.3*: The outcomes of a development-with-offsets scenario relative to a fixed baseline of the initial level of biodiversity in the system under decreasing biodiversity for the general model of a hypothetical ecosystem. The curves show variation in parameters (see Methods for parameter definitions) used to specify the biodiversity trajectory, k_1 (solid lines) and ratio of biodiversity value added to amount lost from the system as a result of development, m (dashed lines). When parameter m<1 offsets are partially implemented (i.e., non-compliance), when m>1 offset multipliers are used. No metric is specified here for biodiversity, so no scale is given, but biodiversity increases above the origin.*



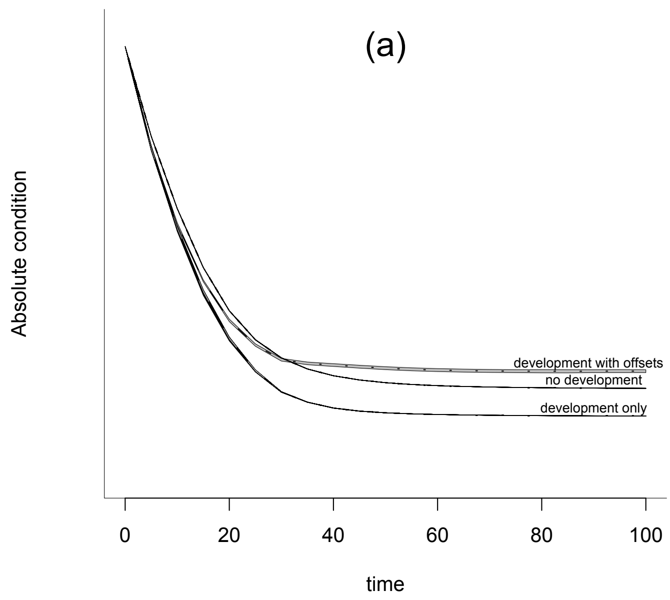
4.3.2 *Real world simulation model*

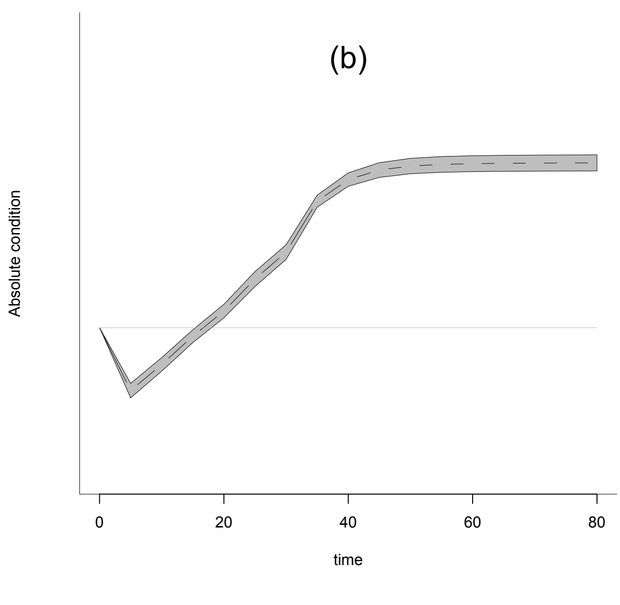
The Melbourne model predicted that development with offsets would result in NL against a fixed baseline, but would be an improvement upon the no-development scenario with decreasing biodiversity (Fig. 4a). This is consistent with the general model (Fig. 4.1a). However, because the Melbourne model was more realistic in including a time lag between development and the maturation of biodiversity gains in offset projects, the improvement only manifested after ~ 30 years. The length of the time lag and absolute value of NL reached over time depended on the biodiversity trajectory.

When I introduced the constraint that biodiversity within offset sites remained stable rather than increasing under management, as was the case for the general model, the Melbourne model resulted in NL against a fixed baseline. In the comparable scenario and frame of reference, the general model predicted that NNL would be achieved because of the assumed lack of time lag. These differences between the general and more realistic models emphasize that offsets may be less effective in meeting objectives once ecological limitations are considered.

Performance of offsetting at the project scale was markedly different than at the landscape scale (Fig. 4.4b). At a project scale, because biodiversity losses occurring in the region outside of areas directly managed under the offset scheme were ignored, the offset policy resulted in a NG from approximately 20 years onward (due to time lag in offset biodiversity gains maturing), against a fixed baseline. The abrupt minimum in the curve, at approximately 5 years, was related to the point at which offsets first begin delivering gains. Specifying a counterfactual at the project scale was not possible because this frame of reference only included areas of grassland when they became either developed or managed as offsets. Thus, these areas were not defined under a no-development counterfactual, and the relative development-only curve would have been the same as the development losses curve.

Figure 4.4: *Results of simulation model of offsets for development impacts upon native Melbourne grasslands: (a) summed grassland condition scores (i.e., biodiversity value) at a landscape scale for a realistically decreasing biodiversity trajectory relative to fixed baseline of overall grassland condition in the landscape at the beginning of the simulation and (b) summed grassland condition under a development-with-offsets scenario for a realistically decreasing biodiversity trajectory relative to a fixed baseline but at a project scale (i.e., representing just the development and offset sites rather than the landscape as a whole). In (b) shaded area represents variation between simulations. Both graphs show mean behavior of 25 simulations, where dashed lines represent the mean and width of each line is the standard deviation.*





4.4 Discussion

4.4.1 *Measuring the success of conservation interventions*

The choice of reference frame for assessing performance affected the apparent success of an offset policy, even when there was no difference in absolute outcomes for biodiversity – the distinction was rather between perceived gains and losses. However, setting different frames of reference at the outset may affect how participants in the scheme behave. The choice of reference frame will affect the actions required to achieve an NNL target. Depending on how this is translated into policy, it may also affect the incentives for land managers and the degree to which they bear the cost of conservation. Land managers may carry out only the minimum restoration required for an offset, particularly if they work for commercial companies motivated by legislation rather than conservation. Against the development-only counterfactual, any minimal effort resulted in NG. But against a fixed baseline and decreasing biodiversity, achieving NNL required more biodiversity gains to be generated from offsetting than was lost from development. In this case, managers would bear the cost of providing biodiversity conservation, whilst society benefits from development but sacrifices no natural capital. However, against the same baseline with increasing biodiversity, a manager could provide no conservation funding and achieve NNL and society would lose an opportunity for natural capital gains.

An intuitive argument for counterfactuals is that they account for the dynamic nature of socio-ecological systems (Nicholson et al., 2009). However, challenges exist in setting counterfactuals, not least in terms of developing and validating a projected trend in biodiversity and anthropogenic impacts when knowledge is poor. Uncertainty is a key barrier to defining counterfactuals (TEEB, 2010). Even fixed baselines are subject to measurement error and knowledge limits (Regan et al., 2002). Using a counterfactual requires strict criteria for judging predictions about trends and ongoing monitoring that continually revisits predictions to test their validity (Ferraro & Pattanayak, 2006; Quétier & Lavorel, 2012). These problems have been identified in relation to a range of conservation interventions such as REDD projects (Angelson, 2008). If future trends are uncertain, there is also an incentive to cheat by overestimating or underestimating future biodiversity decrease to change the amount of offsetting required, depending on which stakeholder group is setting the frame of reference. As such, avoided-loss offsets (where the prevention of future, anticipated, biodiversity losses is considered a conservation gain) can make conservation practitioners, and scientists, uneasy.

4.4.2 *Achieving no net loss against different frames of reference*

Against a fixed baseline at a landscape scale for a region with decreasing biodiversity, I found that ensuring NNL before the end of my simulations required multipliers. However, NG could be expected in the short term even with low or no multipliers if performance was evaluated against a no-development counterfactual. This result suggests that offsets are best equipped to meet certain NNL objectives for regions with decreasing biodiversity.

With increasing biodiversity, even best case offsets performed worse than simply preventing development, making offsets a less preferable option from a conservation perspective (Fig. 4.1c). An implication of this is that the biodiversity trajectory and choice of reference frame could influence whether an offset policy is the best policy option for biodiversity conservation. Offsets still outperform the development-only scenario, but conservationists may not generally consider this an acceptable counterfactual.

The Melbourne model demonstrated the importance of time lags. If lags are taken into account when evaluating policy performance, then multipliers may be insufficient to ensure NNL (Moilanen et al., 2009). One option to resolve this would be to require a biodiversity banking mechanism in which offsets matured in advance of development (Bekessy et al., 2010). My results support the idea that consideration of uncertainty is important in general for planning conservation interventions (Langford et al., 2011). Further, the fact that varying the offset multiplier m between 0.5 - 1.0 (where $m < 1$ suggests a proportion of offsets fail) led to divergent outcomes in the long term suggests that offsets are highly sensitive to even low levels of non-compliance. Thus issues around compliance might be more important then scientific knowledge about the target ecosystem, and compliance is a challenge in even the most established biodiversity offset policies (Gibbons & Lindenmayer, 2007; Chapter 2). This suggests that plans for ongoing monitoring and management of non-compliance should be a pre-requisite for an offset policy.

4.4.3 *Scale of conservation mechanisms*

The models showed that when a frame of reference was set at a project scale, an NG is always possible, assuming there is full compliance with the offset policy, although there may be a time lag. But if offset providers claim to achieve NNL in the absence of clear definitions of scale and reference frame, stakeholders may reasonably be expecting NNL at the landscape scale. As I have intimated, offsets can only support the delivery of NNL in a deteriorating landscape with a fixed baseline at a landscape scale as part of a broader suite of conservation mechanisms.

Alternatively, a landscape scale NNL requirement may appear achievable if a regional offset policy generates substantial leakage of development outside the region (eftec, 2010). That is, offsets could merely displace development activities into other regions not subject to the policy. In this case, the scheme could appear to have achieved NNL relative to even a landscape scale frame of reference, despite major development impacts having occurred elsewhere. These complexities suggest that the scale at which offsets are assessed should be carefully considered in light of the role development plays as a driver of biodiversity loss within the broader social-ecological system. Offsets could be assessed against multiple scales, although different scales may require different objectives.

The choice of whether to use a fixed or relative frame of reference, and of the spatial and temporal scale at which outcomes are evaluated, is at least partly subjective. The only crucial and unassailable requirement, from the viewpoint of conservation science, is that some kind of frame of reference should be transparently specified and the implications of the choice of frame of reference should be appreciated in advance of the intervention. With this in mind, over the coming Chapter, I begin to develop a frame of reference for the Uzbek case study.

Chapter 5

Creating a frame of reference for conservation interventions in the Ustyurt

“Permit me to ask you then, how can man be directing things, if he not only lacks the capacity to draw up any sort of plan even for a laughably short period of time – well, let’s say for a thousand years or so – but cannot even vouch for his own tomorrow?”

Mikhail Bulgakov (from “The Master and Margarita”, 1967)

“My course is set for an uncharted sea.”

Dante Alighieri (from “The Divine Comedy”, 1321)

5.1 Introduction

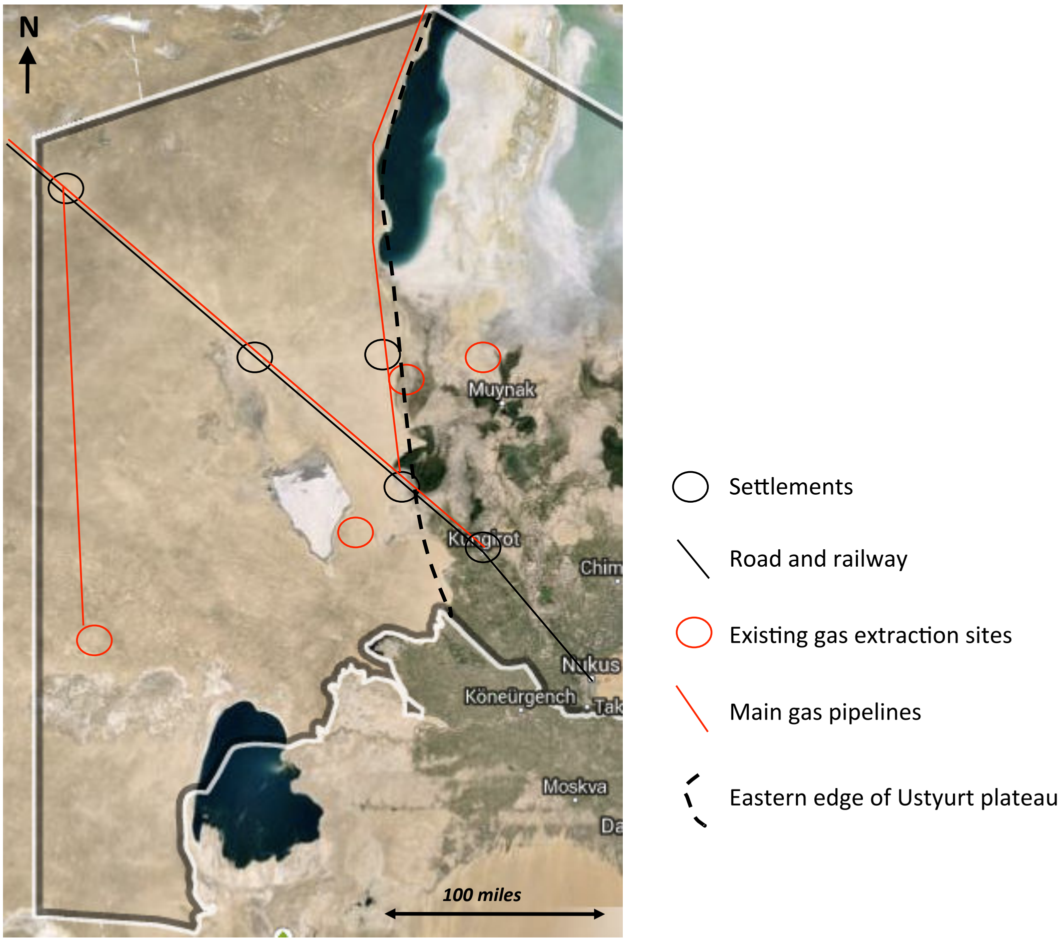
To reiterate central arguments from the previous Chapter: an understanding of the existing context within which conservation interventions take place is critical to effective conservation. The specification of appropriate baselines, which express the current status of a conservation target, would support more rigorous evaluations of conservation success and failures, and thus a more scientific approach to developing conservation policies themselves (Ferraro & Pattanayak, 2006; Maron et al., 2013). However, a baseline understanding of the current status of the target is not adequate in itself. As explored in Chapter 4, there is also a need to project counterfactuals based upon trends, i.e. expectations for what would have occurred in the absence of the intervention (Gordon et al., 2011a; Chapter 4). It is the counterfactual, which can be thought of as a dynamic baseline, that enables measurement of true conservation impact (Ferraro & Pattanayak, 2006). Although counterfactuals are subject to a number of sources of uncertainty (Gordon et al., 2011a), they allow the calculation of the net outcome of interventions rather than merely reporting observed gains (McDonald-Madden et al., 2009).

Baselines and counterfactuals are particularly pertinent in relation to the development of biodiversity offset policies, as a result of the requisite achievement of ‘no net loss’ of biodiversity alongside development (Chapter 2). Few biodiversity offset schemes include the development of both a baseline and a counterfactual as part of a systematic approach to the calculation of true conservation benefit (Quétier & Lavorel, 2012; Maron et al., 2013). I refer to the calculation of a baseline and counterfactual by which to calculate net conservation benefit of an intervention as the development of a ‘frame of reference’ for conservation. A robust frame of reference should not only consider the ecological status quo, but also incorporate physical, social, economic, and institutional factors (Ferraro & Pattanayak, 2006). Further, it is insufficient to consider factors within these domains in isolation, as interactions and feedbacks between them can be important (Nicholson et al., 2009). These factors, both in isolation and in interaction, drive the trajectory of overall biodiversity value in the ecosystem in question (Chapter 4). Finally, a historical perspective is necessary – not only for evaluating the success of potential offset schemes, but also to prevent shifting baseline syndrome (Pauly, 1995), and to provide the social and economic context to which any conservation intervention should be sensitive (Pooley, 2013).

A key reason that counterfactuals are not always developed for conservation interventions is that it is considered difficult to do so, especially where there is inadequate data (TEEB, 2010). Examples do exist of the retrospective evaluation of interventions using a counterfactual, which both emphasize the need for data and show that the use of an appropriate counterfactual change perceived outcomes (Andam et al., 2008), but few examples exist of counterfactuals being developed at the initial intervention design stage. The common outstanding problems with developing counterfactual scenarios for conservation include that it is not done at all, that the assumptions are not made explicit, or that the assumptions made are demonstrably wrong (Maron et al., 2013). In this exploration, I attempt to partially address these obstacles by developing a counterfactual for a case study for which there are very limited data, in which I make my assumptions clear, and in which I compare counterfactuals developed under different assumptions.

The case study is the one outlined in Chapter 3: biodiversity offsets for the residual ecological impacts of oil and gas extraction on the Ustyurt plateau, in Uzbekistan (Fig. 5.1). Again, the feasibility of a biodiversity offsetting policy covering the Ustyurt to compensate is currently being explored (UNDP, 2010a), and the Ustyurt plateau exemplifies how dynamic an ecological and political system can be, and how difficult data can be to obtain (Chapter 3). My approach here is to look at the relatively recent past and identify as far as possible the drivers and patterns of change relevant to management and conservation of the Ustyurt ecosystem. This includes compiling historical datasets and identifying key variables that have been monitored through time.

Figure 5.1: *Satellite map of the Ustyurt plateau in northwest Uzbekistan, showing national boundaries (grey and white line), with approximate locations of settlements, transport infrastructure, gas extraction sites and pipelines. The study area is that bounded by the Uzbek border to the north, west and south, and the edge of the plateau to the east. Satellite image source: Google Inc., 2014.*



Because biodiversity offsets tend to use either habitat-based (floral) or species-based (faunal) metrics to calculate no net loss (Quétier & Lavorel, 2012) I define my conservation targets either as the Ustyurt vegetation (habitat-based metric) or the status of particular species of interest (species-based metric). This study provides insights into the drivers of ecological change for an interesting and relatively neglected region, and highlights some of the practical and theoretical challenges that arise when developing frames of reference for conservation interventions.

5.2 Methods

We set the context with a brief history of the Ustyurt region of Uzbekistan over the last 100 years. This is approximately how long the republic has existed as a defined international entity, albeit originally under Soviet rule.

I gathered information on trends in primary conservation targets in the Ustyurt, categorized into the habitat and species targets. Subsequently, I explored the drivers of ecological change in the region, and developed a conceptual map of the main interactions between these drivers. I explicitly considered the drivers of change in four domains (physical, social, economic, institutional). Finally I used these analyses to develop a frame of reference (a baseline and counterfactual) that could be used to assess the effectiveness of the planned intervention in the region; a biodiversity offset for gas infrastructure. The data were collected and analyzed over a period of 27 months (2010 – 2013), incorporating primary and secondary data sets acquired during three field trips (Gintzburger et al., 2011; Chapter 6), as well as information available online (Table A2.1). The ecological and technical rationales for the methods used are included in the appendices (Appendix 2), and only the trends in ecological status and drivers of change in status are presented in the main text.

5.2.1 *Habitat target: green vegetation cover*

Habitat-based metrics for biodiversity offsetting generally measure area and condition of vegetation (Quétier & Lavorel, 2012). In the Ustyurt, a measurable component of condition important both for rangeland management purposes and for conservation is the amount of green vegetation cover (Opp, 2005; Gintzburger et al., 2011). To gain a landscape scale assessment of trends in green vegetative cover over recent decades, I used remotely sensed data sets, with the Normalized Difference Vegetation Index (NDVI) as my focal metric.

The spring and summer seasons are the time at which vegetation cover is extensive enough to permit use of the NDVI. Bekenov et al. (1998) give seasons in the Ustyurt as: spring (early/mid-March to early June), summer (early/mid-June to early/mid-September), autumn (mid/late September to early November) and winter (November to early/mid-March). These definitions are used throughout. I used spring and summer NDVI from three different satellite data sets to examine vegetation dynamics during the growing season over the period 1982 - 2012 (Robinson et al., 2000; Singh et al. 2010a).

For trends in the distribution of vegetation cover, I created a raster layer of the average spring NDVI values for each year, stacked these raster layers, and completed a linear regression analysis pixel by pixel. This allowed calculation of a gradient for the approximate trend in NDVI values for each pixel for the years in question, in turn permitting the creation of a spatial map of NDVI trends across the region. Standard least squares regression was used to calculate the gradient by pixel, with NDVI as the dependent variable and time as the independent variable.

5.2.2 *Species target: saiga antelope*

Species-based metrics for biodiversity offsets are designed either to maintain or enhance overall abundance of a species itself, or to manage habitat for that specific species. I explored trends in both the abundance and spatial distribution of the region's flagship conservation target species – the saiga antelope *Saiga tatarica*. As explained in Chapter 3, the saiga is Critically Endangered, and only 5 populations remain in the wild (Milner-Gulland, 2010). The Ustyurt plateau population predominantly spends the summer in Kazakhstan but overwinters in northwest Uzbekistan (Bekenov et al., 1998). Abundance data for the saiga population for Uzbekistan and for the Ustyurt plateau as a whole (1980 – 2012) were compiled from reports of aerial and vehicle surveys undertaken by the Institute of Zoology in Kazakhstan, participatory monitoring efforts and sources in the literature (Bekenov et al., 1998; Milner-Gulland et al., 2001; Milner-Gulland, 2010). Abundance data were used to explore trends as a result of direct saiga mortality (e.g. poaching), while distributional data were used to shed light upon changes in habitat use based on anthropogenic (e.g. Singh et al., 2010b) and environmental factors.

5.2.3 *Drivers of change in conservation targets*

Secondary data sets were analysed for trends in those factors hypothesised to affect conservation targets. Where appropriate, relationships between variables were analysed using generalised linear models in the statistical package R v2.15.1 (R Development Core Team, 2012).

<u>Physical drivers</u>

Climate change is potentially a driver of ecological trends in the Ustyurt (Singh et al., 2010a; Chapter 3), so data were obtained from meteorological stations at the Jaslyk and Karakalpakia settlements on the Ustyurt. Data included mean/maximum/minimum temperature, rainfall, snowfall, and snow cover. These are all potentially influential both for vegetation growth and as drivers of saiga distribution and migration. Dust deposition, carried by strong desert winds, is known to have an impact on both ecological systems and the health of the human population in the Ustyurt (Micklin, 2007), hence I also included analyses of wind direction (I. Aslanov, unpublished data). Desertification and status of faunal and floral species are partly determined by water availability, in conjunction with other drivers of change, and and so I included information on water resources where available.

<u>Social drivers</u>

Human population data for the region were obtained from the administrative centre in the town of Nukus. Censuses for Uzbekistan are available online (UN, 2011). Human population trends are relevant to the development of a counterfactual as the number, density and distribution of people will influence natural resource use on the plateau as well as demands on natural resources and the extent of development and infrastructure.

I also tracked unemployment rates, using a combination of official socio-economic data online (Dynamic Lines, 2011) and socioeconomic surveys completed in the region (Bykova & Esipov, 2004; Kühl et al., 2009; Phillipson & Milner-Gulland, 2011). Poaching is known to be the primary cause of saiga decline (Milner-Gulland et al., 2003), and is linked to socioeconomic factors such as poverty and unemployment (Kühl et al., 2009; Phillipson & Milner-Gulland, 2011).

Economic drivers

Agriculture, primarily animal husbandry, is an important regional economic activity. Livestock require water and forage resources, which are potentially limited in this environment, and agriculture is therefore relevant for this study. Similarly, extensive agricultural land use may disturb wild faunal habitat use. Data on livestock and agricultural land-uses from 1990 to 2006 were taken from the report "Livestock raising in Uzbekistan" (UNDP, 2010b), and obtained in hard copy from the Ministry of Agriculture, Karakalpakstan.

The primary industry on the plateau is oil and gas extraction, and biodiversity offsets have been proposed as a means to bring biodiversity into the mainstream of planning by this sector. Industry has a range of direct and indirect negative impacts upon biodiversity in the Ustyurt, including habitat clearance and species disturbance (UNDP, 2010a; Mott Macdonald 2012). I collected field data on the spatial configuration of oil and gas infrastructure on the plateau, and vegetation impacts associated with this infrastructure (Chapter 6).

Institutional drivers

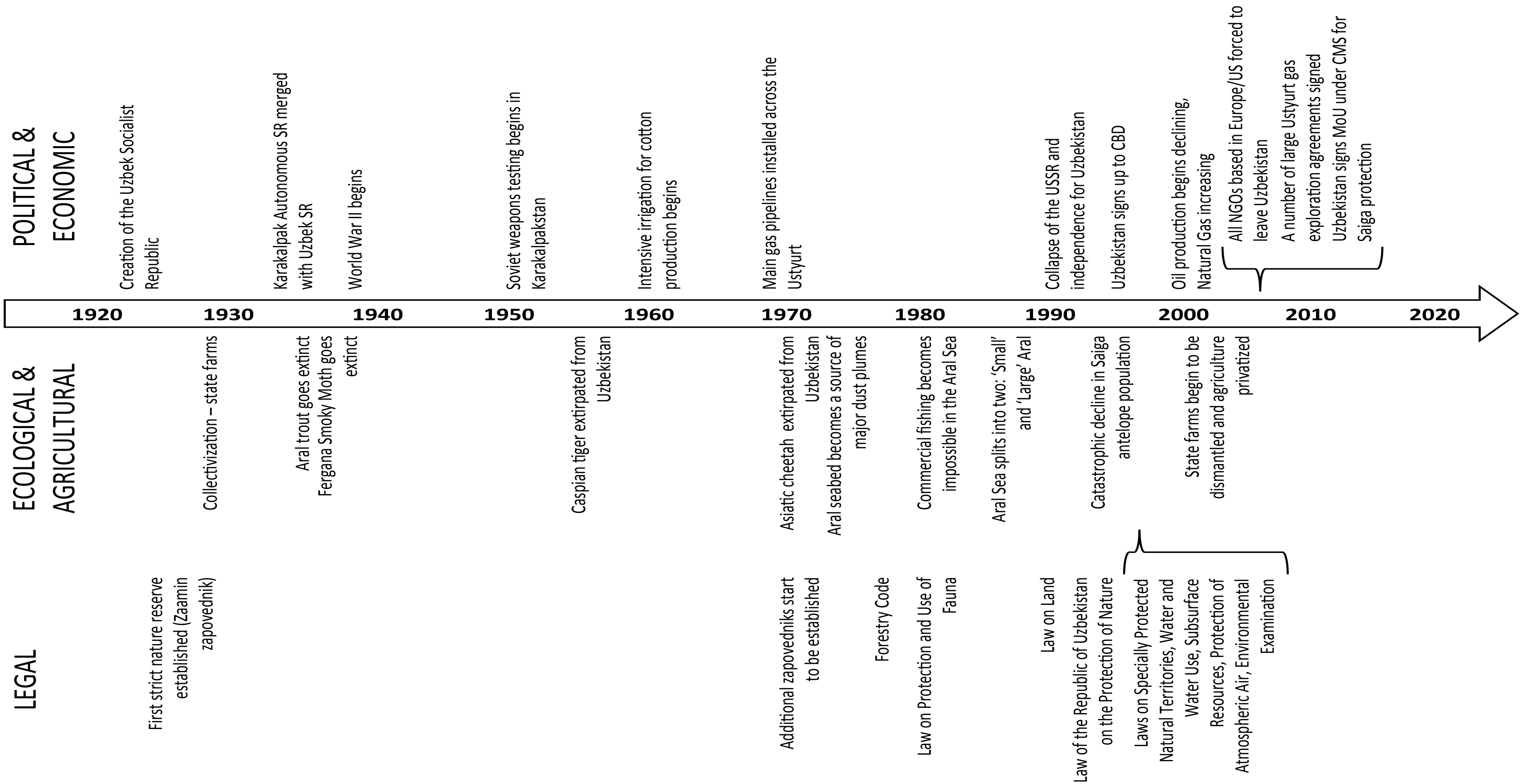
As has already emerged as a theme in this thesis, compliance is crucial to the outcomes of biodiversity offset projects (Chapters 2, 4). I considered the current national legislation with respect to biodiversity conservation and, in particular, legislation that could facilitate offsetting. Further, I evaluated the administrative structure available to manage any offset scheme, as well as the influence of non-governmental organizations. I also explored where possible the land tenure system, which determines land ownership and use rights (Robinson et al., 2013).

5.3 Results

5.3.1 *A brief recent history of the Ustyurt*

The last century has been a time of political, social and environmental upheaval in the Ustyurt (Fig. 5.2; Micklin, 2007; Asimov et al., 2009; UNDP 2010a; Robinson et al., 2013). The 1920s saw the creation of the Uzbek Socialist Republic (SR), the first time that a republic with those borders categorized as 'Uzbek' had existed. The decade saw the collectivisation of farms across the Soviet Union with ramifications for farming practices and land tenure in Uzbekistan to this day, but also the creation of the earliest national nature reserve. In the 1930s the area now known as Karakalpakstan, containing the Ustyurt plateau and part of the Aral Sea, was merged with the Uzbek SR. The Aral Sea was clearly being heavily fished at this time, as the endemic Aral Trout was last recorded in the 1930s (Asimov et al., 2009). As part of the Soviet Union Uzbekistan was pulled into World War II to provide resources, and the remote Ustyurt was used as a weapons testing facility – an activity that intensified in the 1950s, and on into the Cold War.

***Figure 5.2**: Key events in relation to contemporary conservation efforts over the last 100 years in the Ustyurt*



The background loss of charismatic species continued with the extirpation of the Caspian Tiger in the 1950s (Asimov et al., 2009). A policy with severe ramifications for conservation in Karakalpakstan was implemented in the 1960s: the use of widespread irrigation for cotton along the banks of the Amu Darya, which by the early 1980s caused the Aral fishing industry to collapse entirely, and by the late 1980s reduced the extent of the Aral so much that it split into two smaller lakes (Micklin, 2007). The 1960s also saw a boom in extractive activity in and around the Ustyurt plateau. In the early 1970s, large trunk gas pipelines commenced construction. At this stage a suite of nature reserves were designated, but the Asiatic cheetah was still extirpated in the 1970s. In 1991, the Soviet Union collapsed, precipitating independence for Uzbekistan for the first time. The following two decades saw a proliferation of environmental legislation alongside increasing natural gas activity in the Ustyurt, and the catastrophic decline of the saiga antelope population, which has consequently become one of the flagship species for conservation in the Ustyurt. In the early 2000s, the state farms began to be dismantled (Robinson et al., 2012), but land has yet to be privatised. The mid 2000s were particularly notable for a well-publicized incident in the town of Andijan in the south of Uzbekistan, the response to which resulted in all external NGOs being required to leave the country – a situation that has not changed to this day.

5.3.2 *Trends in conservation targets*

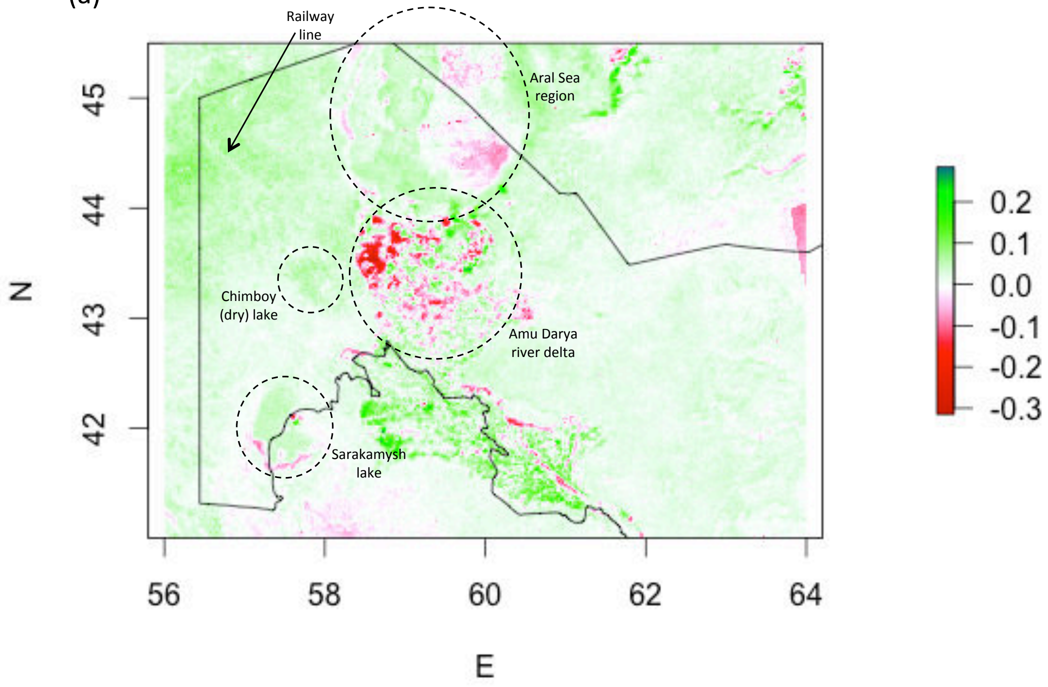
Habitat

The annual NDVI pattern tends to a minimum over winter and a maximum around May, with a drop in July/August and secondary peak in September (Fig. A2.2). This coincides with the pattern of actual vegetation cover that might be expected for the northern hemisphere, and which can be observed in the field. No decadal trends were apparent in mean NDVI across the Ustyurt during the period 1982 – 2006 (Fig. A2.3).

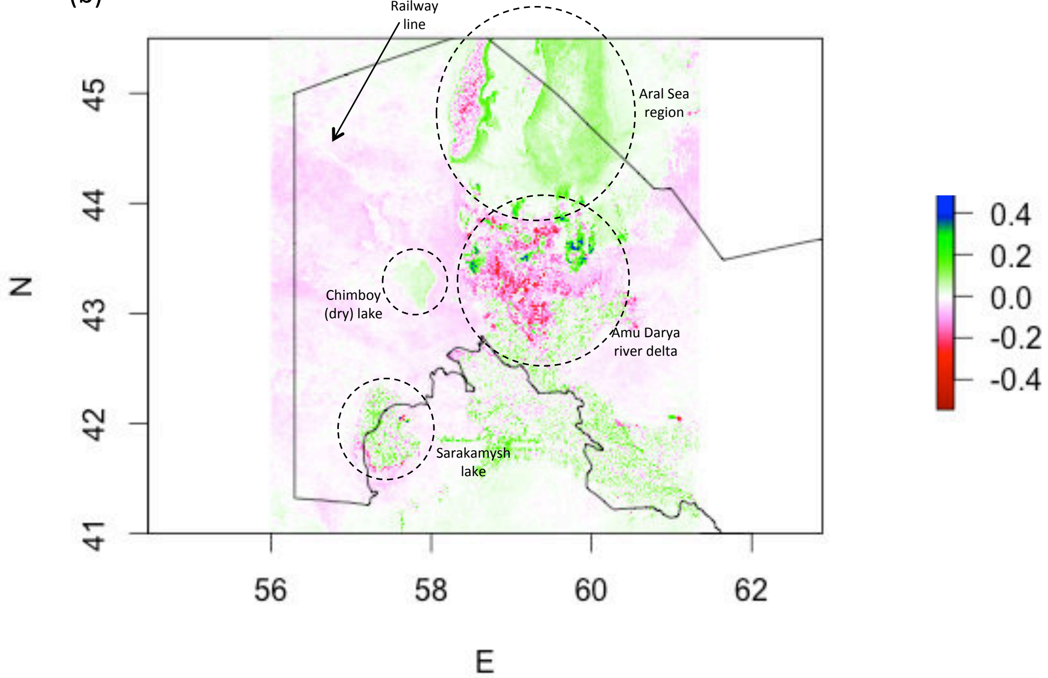
Changes in the distribution of NDVI can be observed at a finer scale, however, and are indicative of spatially heterogeneous habitat change (Fig. 5.3). Areas of either dramatic increases or decreases in NDVI correspond with irrigated agricultural activities, both in the Amu Darya river delta and Aral Sea region. As is the case here, NDVI can be used to highlight those areas in which changes in vegetation are occurring most rapidly and in which green vegetative cover seems to be decreasing, which I interpret as being those that require the most urgent conservation attention. Note that caution should be taken in interpreting changes in the NDVI value of a pixel, as the index is based upon reflected light rather than a direct measure of vegetation cover or condition.

Figure 5.3*: Absolute trend in spatial distribution of mean spring NDVI (dimensionless) based upon linear regression per pixel (a) in 1991 – 2003, using a data set from the Kansas Applied Remote Sensing Program, (b) in 2001 – 2012, using MODIS data. Positive = increase in NDVI, negative = decrease in NDVI. The two plots use a vegetation index that is calculated using the same method and shown using the same projection, but with data from different satellites – so whilst comparable, the two sets of data were plotted separately rather than on one continuous timeline.*

(a)



(b)



Upon visual inspection, NDVI across the Uzbek Ustyurt generally increased from 1991 to 2003, but remained stable or decreased from 2001 to 2012 (Fig. 5.3). It is not clear if the increase over the first decade would have been related to climatic factors, or a change in land use following the collapse of the Soviet Union, or some other factor. The Aral Sea region appeared to increase in NDVI more recently, which may be due to the gradual vegetation of the exposed seabed. The Amu Darya river delta is characterised by patches of both steep increase and decrease in vegetation, perhaps reflecting irrigated agriculture in the area. A straight line is partially visible cutting across the NW plateau where NDVI has remained stable since 1991, corresponding to the railway, asphalt road, and main trunk gas pipeline. Close to this line, NDVI has trended upwards in 1991 – 2003, and then downwards in 2001 – 2012. In the far south of the plateau, NDVI appears to have remained relatively more stable than further north, apart form one area which corresponds to the saline lake Sarakamysh, where NDVI has generally trended upwards.

Over the last decade, then, the Ustyurt plateau has been characterised by a heterogeneous decrease in NDVI, which is consistent with the suggestion that desertification is a concern for the region (Opp, 2005). However, over a longer time period of a few decades, the case could be made that NDVI, and therefore vegetation cover, has not shown a clear trend.

It is worthy of note that, throughout the Uzbek Ustyurt, the lichen *Tortula desertorum* can be found. The widespread presence of this lichen is associated with low grazing pressure from domesticated or wild animals. This could become a concern if it prevents recruitment of new scrub (Gintzburger et al., 2003; Esipov & Shomudurov, 2011), and in this case might provide an alternative indicator of habitat degradation.

Species

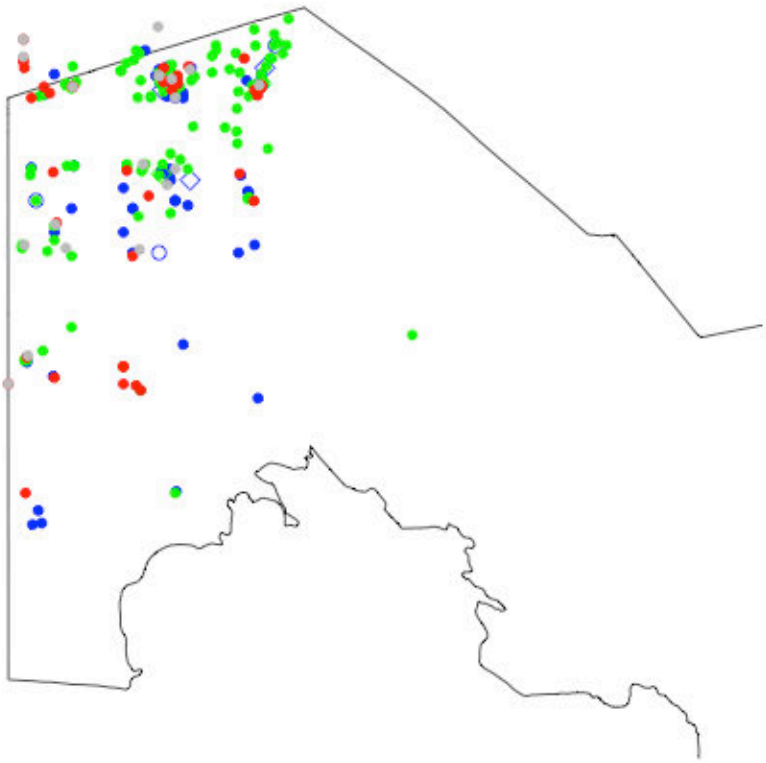
The fauna of Uzbekistan includes numerous species of conservation concern, as discussed in detail in the National Red List for Uzbekistan (Asimov et al., 2009). The sequence of extirpations and extinctions over the last 100 years (Fig. 5.2) represents a trend of decline in charismatic vertebrate and other species in the Ustyurt.

Turning to the flagship species of the region, the Ustyurt saiga population as a whole experienced a crash in the late 1990s and early 2000s (Fig. A2.4; Chapter 3). Over the last century, and into the present day, the range of the saiga population in Uzbekistan has remained approximately consistent (Bekenov et al., 2009). The annual saiga migration south is, in part, triggered by temperature or some threshold snow depth (Esipov et al., 2009), which suggests that any upward trend in winter temperature over time could influence how far south into Uzbekistan the species will migrate. An unknown and variable proportion of the population enter Uzbekistan every year. High saiga mortality is recorded in the literature in association with severe winters in the late 1980s and early 1990s (Esipov et al., 2009), but not reflected in broad-scale population data. Indeed, as no consistent or comparable trans-boundary monitoring takes place, it is difficult to know how well the available data for the population as a whole (including Kazakhstan) reflect the situation for saigas in Uzbekistan, beyond that there has been a recent and substantial decline.

A recent hurdle to the persistence of the Ustyurt saiga population is the 2013 construction of a boundary fence by Kazakhstan along the Kazakh-Uzbek border, which largely cuts off the saiga migration route (Salemgareyev, 2013). The fence may have severe implications when the population migrates to avoid harsh winters (Milner-Gulland, 2012). GPS collaring data have shown individual saiga moving alongside the fence and eventually passing through, suggesting that it obstructs movement but is not impassable (Salemgareyev, 2013).

Recently collected monitoring data do suggest the possibility of small, permanent saiga populations resident in Uzbekistan i.e. those that remain during the summer months (Fig. 5.4).

Figure 5.4*: Locations of saiga observations, according to all available participatory monitoring data, transect data and general observations from 2006 – 2012. Blue = winter, green = spring, red = summer, grey = autumn. If 'n' is the number of saiga observed in the herd, then for full circles n<500, for empty circles 500<n<1000, and for empty diamonds n>1000.*



5.3.3 *Trends in physical drivers*

Climate

Monthly mean temperatures over the period 1977 - 2005 show little variation between years

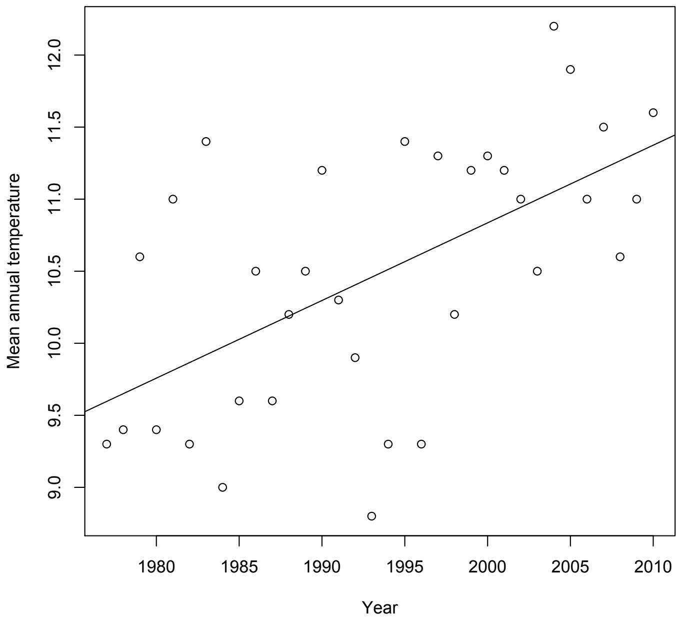
(Fig. A2.5). Average temperature peaks in July at 28.1 ± 0.5 ^{0}C, dropping to -5.6 ± 1.0 ^{0}C in January. Peak monthly rainfall (15.1 ± 5.6 mm/month) coincides with peak NDVI in May. There is significantly more inter-annual variation in rainfall than temperature (Fig. A2.6).

Temperature:

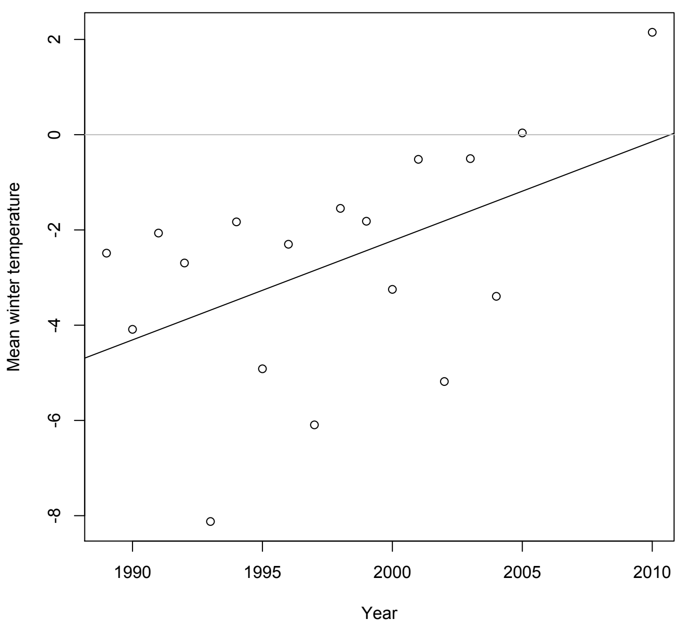
Mean temperature has trended upwards at the Jaslyk meteorological station since 1977 (Fig. 5.5a). Data are available for meteorological stations in other settlements on the Ustyurt plateau for 2005 - 2010. In these years, the temperature data between settlements show good agreement (Fig. A2.7). As such, there are grounds for arguing that the Jaslyk data are representative of the plateau as a whole. These data do not necessarily indicate a longer term trend in temperature, but conform with the IPCC data for the Central Asia region, which records a historic temperature rise of 1 – 2 ^{0}C per annum (Meehl et al., 2007) and with other analyses of temperature trends across Uzbekistan (Lioubimtseva & Cole, 2006; Ackura et al., 2008). The recent rise in annual mean temperature was accompanied by an increase in mean winter temperature (Fig. 5.5b).

Figure 5.5*: (a) Mean annual temperature at Jaslyk meteorological station, 1977 – 2010 (linear model: r^2 = 0.340, p = 0.0003). (b) mean winter temperature at Jaslyk, 1977 – 2010 (linear model: r^2 = 0.249, p = 0.035).*

(a)



(b)



Precipitation:

No significant trend was found in the Jaslyk rainfall data from 1977 to 2010. Annual rainfall was variable, both in terms of absolute rainfall and its variability. This is consistent both with Ackura et al. (2008) and IPCC climate predictions for Central Asia (Meehl et al., 2007). Particularly high snowfall was experienced in the late 1980s and early 1990s. Alongside an upwards trend in winter temperature (Fig. 5.5), there was relatively low snowfall from the mid 1990s onwards (average sum snowfall from 1985 – 1995 was = 514.1 ± 466.5 mm; from 1995 – 2005 was = 72 ± 30.1 mm).

Wind:

There is a highly dominant easterly or northeasterly wind across the Ustyurt (Fig. A2.8). Large-scale dust deposition, from the desiccated Aral Sea to the northeast of the plateau, is known to have acute impacts upon biological receptors (Micklin, 2007). No localised impacts upon vegetation from dust deposition associated with traffic, in the dominant wind direction, were noted in the field (Chapter 6).

Desertification

Desertification is amongst the main environmental problems facing Uzbekistan (UN, 2010), in conjunction with soil salinisation, particularly in the region adjacent to the Aral Sea (Ji, 2008; Micklin, 2007). A gradual depletion of water resources in northwest Uzbekistan (Akcura et al., 2008) is part of this trend. The drying of the Aral Sea is a highly visible outcome of this depletion, resulting from the diversion of water from the Amu Darya and Syr Darya rivers for crop irrigation (Micklin, 2007).

A network of wells is in place across the plateau, and those near settlements and the asphalt road are heavily used. There are approximately 300 wells in the Karakalpakstan portion of the Ustyurt plateau (Gintzburger et al., 2011). Many in remote areas are no longer operational, consistent with deterioration of infrastructure for livestock since the fall of the Soviet Union (Robinson et al., 2013). The trend in surface salinisation in the Ustyurt accompanies a trend towards salinisation of underground water resources (Ackura et al., 2008). The combination of salinisation and restricted locations of functioning wells (Robinson et al., 2013) means that water could become a concern for livestock raising and agriculture. By extension, this may also be an increasing problem for wild flora and fauna. It should be noted that extensive water abstraction is a requirement of extractive industry.

Interactions between climate and vegetation

A significant amount of the variation in mean spring and summer NDVI can be explained by mean temperature and summed rainfall from the previous month (Table A2.2). This lagged relationship is consistent with the literature (e.g. Robinson, 2000), and confirms the assertion made in Gintzburger et al. (2003) that temperature and water availability are important factors in shaping species associations found in this habitat. Mean and minimum temperatures are also positively correlated with greater landscape scale variation in NDVI values.

Interactions between climate, vegetation and saiga

Singh et al. (2010a) find that the spatial distribution of saiga populations is driven by climate and forage availability, but that these drivers are relatively weak for the Ustyurt population. Due to the extremely low numbers of saiga even in comparison to two decades ago (Fig. A2.4), forage is unlikely to be a limiting factor.

5.3.4 *Social trends*

Human population

The human population of Uzbekistan has been growing since the 1960s, and is projected to stabilize around 2050 (UN, 2011). This trend is not evident in the Ustyurt region. Official unemployment rates in Uzbekistan are high but the data are uncertain. Official statistics put unemployment at 32% in 2010 for the country as a whole (up from 20% in 1991; Dynamic Lines, 2011). Conversely, a recent socioeconomic study reported an official figure of 25% unemployment in settlements in Karakalpakstan, and that 29% of households surveyed in the Uzbek Ustyurt “contained unemployed inhabitants” (Phillipson & Milner-Gulland, 2011). Roughly 25% of households surveyed reported that they were in the process of relocating to Kazakhstan due to the promise of better living conditions (Phillipson & Milner-Gulland, 2011). The ratio of Uzbek to immigrant workers in the Ustyurt is unclear.

Interaction between social and ecological trends

Unemployment and livelihood trends are important in terms of biodiversity conservation, as low household income continues to be the main driver of poaching activity in the Uzbek Ustyurt (Bykova & Esipov, 2004; Phillipson & Milner-Gulland, 2011).

5.3.5 *Economic trends*

Livestock numbers

On the Ustyurt plateau, the agricultural sector is primarily pastoralist (UNDP, 2010b). In 2007, the UNDP recorded 3,188,800 ha of agricultural land in the Karakalpak republic, of which only 415,700 ha was arable land and the majority of the remainder was pastures. Data on livestock numbers in 2010 are available for the districts of Muynak and Kungrad, which include the Ustyurt plateau (Fig. A2.9). Detailed data on the proportion or location of livestock kept on the plateau itself are not publically available; however, cattle are largely kept in the Amu Darya river basin to the east of the Ustyurt and graze along riparian habitats, whereas livestock kept on the plateau are primarily sheep, goats and camels (J. Bull, pers. obs.).

Cattle numbers in Karakalpakstan in 1990 – 2010 appeared stable, with an absolute increase in recent years (Fig. A2.10). Although the majority of cattle are not kept on the plateau, this gives an indication of trends in animal husbandry, consistent with other reports that have discussed livestock in post-Soviet Uzbekistan (Gintzburger et al., 2003; Robinson et al, 2013).

Interactions between livestock and vegetation

Low levels of vegetation cover and the presence of species indicating rangelands degraded by grazing are evident near settlements on the Ustyurt, along the main roads, and along the network of wells (Gintzburger et al., 2011). Near settlements, the vegetation is largely cleared within a perimeter of 8 – 9 km, approximately the distance that sheep can travel in one day and return for water in the evening (Gintzburger et al., 2011). However, there are few settlements on the plateau, so there is a limited impact overall.

Industry

The main commercial activities on the Ustyurt are extractive industries. These have negative impacts upon regional biodiversity (UNDP, 2010a; Mott Macdonald, 2012) that have yet to be fully quantified – although this will be elaborated upon in Chapter 6. Uzbekistan is also the fifth largest exporter of cotton in the world (NCC, 2012), and the cotton industry is linked to the partial loss of the Aral Sea. This loss has itself directly influenced two other industries in the area: oil and gas, and fishing: there are large gas reserves under the Aral Sea, so the retreat of the shoreline is facilitating access and extraction (EIA, 2012), and the fishing industry has collapsed.

The oil and gas (O&G) industry in Uzbekistan is large and increasing. Further details on this industry in Uzbekistan and Karakalpakstan are contained in Chapter 3 and UNDP (2010a). Official government statistics since independence in 1991 show mixed trends. Oil production in Uzbekistan has declined in recent years in part due to the realization of significant gas reserves in Karakalpakstan, and subsequent diversion of resources towards gas exploration/extraction (EIA, 2012). Gas production, conversely, has trended upwards since the early 1990s, (EIA, 2012; Fig. A2.13), and a number of significant gas developments are planned (e.g. Mott Macdonald, 2012).

O&G infrastructure is distributed throughout the Ustyurt (Fig. 5.1; Fig. A2.12). Uzbekistan as a whole has over 10,000 km of oil and gas pipelines, one of the highest for any country in the world (Goodland, 2005), the primary one being the Bukhara–Tashkent–Bishkek–Almaty (BTBA) pipeline that passes through the Uzbek Ustyurt from the southeast, and on into southern Kazakhstan (Yenikeyeff, 2008). In addition, the asphalt road and rail crossing the Ustyurt plateau parallel with the BTBA pipeline are used by the industry, and link two of the main settlements in the region: Jaslyk and Karakalpakia. All three main pipeline routes are currently undergoing some form of maintenance, improvement or expansion to meet increasing production capacity (J. Bull, pers. obs.). Ustyurt infrastructure is mapped in Chapter 6.

Interaction between industry, socioeconomics and conservation targets

As will be evidenced in Chapter 6, development and infrastructure remove vegetation almost entirely within the space they directly occupy, but impacts upon vegetation attenuate within 25 m. The effect of settlement creation or expansion for workers would have a broader footprint, due to the practice of removal of most vegetative cover through grazing for 6 – 8 km.

Unemployment and low income has been strongly linked to saiga poaching in the region, as a result of the market value for saiga horn (Phillipson & Milner-Gulland, 2011). A growing natural gas industry in the region means increased work opportunities, if employment is given to resident workers rather than migrants. This interaction presents a good example of why feedback loops within an ecosystem should be considered in conservation schemes: industrial development in the region is exacerbating vegetation loss (Chapter 6), and potentially causing some direct mortality of threatened species (Mott Macdonald, 2012) amongst other environmental impacts (UNDP, 2010a). However, at the same time, industry could provide at least some employment in a region where unemployment is known to drive poaching of one of the main conservation targets. On the other hand, poaching for meat could increase with an influx of new employees and their families, either by locals for sale to industrial employees, or by the workers and their families themselves. This means that if development preferentially employed local people it may contribute towards conservation solutions in the region; if not, it may exacerbate existing problems.

Conversely, anthropogenic presence is also known to disturb saigas (Singh et al., 2010b; UNDP, 2010a) and vertebrates in general (Benitez-Lopez et al., 2010); furthermore, associations have been made between the presence of industry and direct saiga mortality, in trenches for example (Mott Macdonald, 2012) although this has not been quantified. As industry expands further across the Ustyurt, it is likely to come into further conflict with the remaining saigas.

5.3.6 *Institutional context*

In this section, I focus on the institutional and legislative context relevant to biodiversity conservation, oil and gas, and biodiversity offsetting in Uzbekistan.

All institutions for the management and conservation of natural resources in Uzbekistan have emerged since independence in 1991. The primary state institution for biodiversity conservation in Uzbekistan is the State Committee for Nature Protection (*Goskompriroda*). Notable amongst its subsidiaries are *Gosbiokontrol*, responsible for managing protected areas, hunting and anti-poaching, and *Glavcosecoexpertiza*, which carries out environmental and ecological impact assessment. Smaller local replicate institutions carry out these same tasks for the region of Karakalpakstan, in partnership with the main state body in Tashkent. Much of the study area is managed by the Ministry of Agriculture, Forestry and Fisheries for natural resources, and this same Ministry would undertake any re-vegetation or habitat restoration activities required as part of an offset scheme.

All issues relating to oil and gas in Uzbekistan are managed through the state organization (*Uzbekneftegaz*), which partners with international organizations to undertake gas extraction projects (e.g. KoGas, LukOil, Gazprom, CNPC). Two key natural gas extraction sites in the Ustyurt region are Surgil (KoGas) and Shakpahkty (Gazprom). *Uzbekneftegaz* has a research subsidiary, which would be involved in developing new methodologies for implementation in the sector, such as biodiversity offsetting (*Uzlitineftegaz*). On the academic research side, the Uzbek Academy of Sciences until recently undertook ecological research through the separate Institutes of Zoology and Botany, although these two have now merged into a larger institute.

Beyond the state organizations for nature protection and management, there are some third and public sector conservation organizations supporting operations in the region. These include international organizations such as the United Nations Development Programme (UNDP, 2010a) and Japanese International Cooperation Agency (JICA, 2011). There are no international NGOs registered in Uzbekistan, but some provide support for activities in the region, such as the Saiga Conservation Alliance (SCA, 2012).

Existing environmental legislation makes provision for the rational and sustainable use of resources such as forestry and soils, the regulation of water abstraction and use, the protection of specific flora, fauna and habitats, the prevention and management of pollutants to soil air and water, and the examination of environmental impacts (including requirements upon Environmental Impact Assessment for industrial developments: Cabinet of Ministers, 2001; UNDP, 2010a).

An established network of protected areas exists across the country, divided into a hierarchy aligned with the established IUCN categories for protection. 'Zapovedniks' are State Strict Nature Reserves (IUCN category I); State National Parks and Nurseries for Rare Animals correspond to IUCN category II and III respectively. Also important are 'zakazniks' (i.e. State Reserves) that are equivalent to IUCN category IV. Experience of implementing protected areas is extensive, the oldest protected area in the country being the Zaamin Mountain zapovednik, which was established in the 1920s (Fig. 5.2). In the Ustyurt plateau, the only existing protected area is the 1991 Saigachy zakaznik, but this has neither staff nor budget and so is ineffective in protecting the threatened species it contains (Esipov et al., 2009).

Interaction between economic and institutional factors

Livestock on the Ustyurt are currently under a mixture of state and private ownership (including mixed flocks), although farmers do not generally own land aside from some small household plots (Lerman, 2008). The vast majority of land is owned by the state, as was the case during the Soviet era. There is supposed to be a system of charging for grazing access, but this is only partially implemented (Robinson et al., 2013). The relevance of the land tenure situation to this discussion is twofold: firstly, land tenure influences grazing regimes, which as described have an impact upon the condition of habitat. Secondly, tenure is of importance with regards to designing and implementing biodiversity offset schemes (Gordon et al., 2011b), affecting who has to pay or be paid to maintain restoration areas, and affecting the nature of legal agreements that ensure ongoing management of offset projects.

Discussions with Uzbek legal experts revealed that land can be effectively rented by private sector organisations seeking to explore and extract O&G – there is a legal obligation to retain environmental characteristics on such land, but no explicit guidance for biodiversity. Land can be temporarily rented for up to 10 years, or more permanently held, but is never owned. This means there is a lack of opportunity for private landowners to provide offset receptor sites, and for a private market in biodiversity credits. Whilst some speculate about reform to the land ownership system (Robinson et al., 2013), those consulted in Uzbekistan thought it unlikely that reform would take place in the short-term future.

There is an established framework of environmental legislation in place, and Uzbekistan is signatory to both the Convention on Biological Diversity and the Convention on Migratory Species. It is also a signatory to a Memorandum of Understanding and Action Plan on saiga conservation under the CMS, and to a bilateral agreement on saiga conservation with Kazakhstan. The legislation relevant to the implementation of an initial biodiversity offset policy in Karakalpakstan currently contains gaps. These is a requirement to evaluate and monitor the ecological impacts of industrial activities, and a no net loss compensation requirement (UNDP, 2010a). However, institutional capacity already exists to provide a basis for offsetting, particularly in relation to creating protected areas and managing the environmental impacts of industry. Furthermore, it is clear that offsetting may play an important part in conservation in the region, given that industrial expansion will happen, and that economic development may be a pre-requisite for effective conservation.

There currently exist at least three key institutional barriers to robust biodiversity offsetting: a lack of expertise in bringing the topic of biodiversity into the environmental impact assessment process; insufficient resources or capabilities to ensure compliance and enforce regulations in Karakalpakstan (cf Chapter 2); and, a lack of independent organizations, such as environmental NGOs, to monitor offsetting activities.

5.4 The frame of reference

Having considered elements of the case study in isolation, I now attempt to combine them into a useful frame of reference. The data sets compiled here and in Appendix 2 constitute the baseline for the case study – one of a landscape containing large areas of relatively intact habitat, with populations of typical Central Asian steppe, desert and riparian species. There are few endemics, because of the contiguous nature of the region's arid ecosystems. Some species are at very low numbers or extirpated, within and adjacent to the now highly degraded Aral Sea. The area contains a sparse but established human population and industrial infrastructure (Table 5.1).

5.4.1 *Interactions*

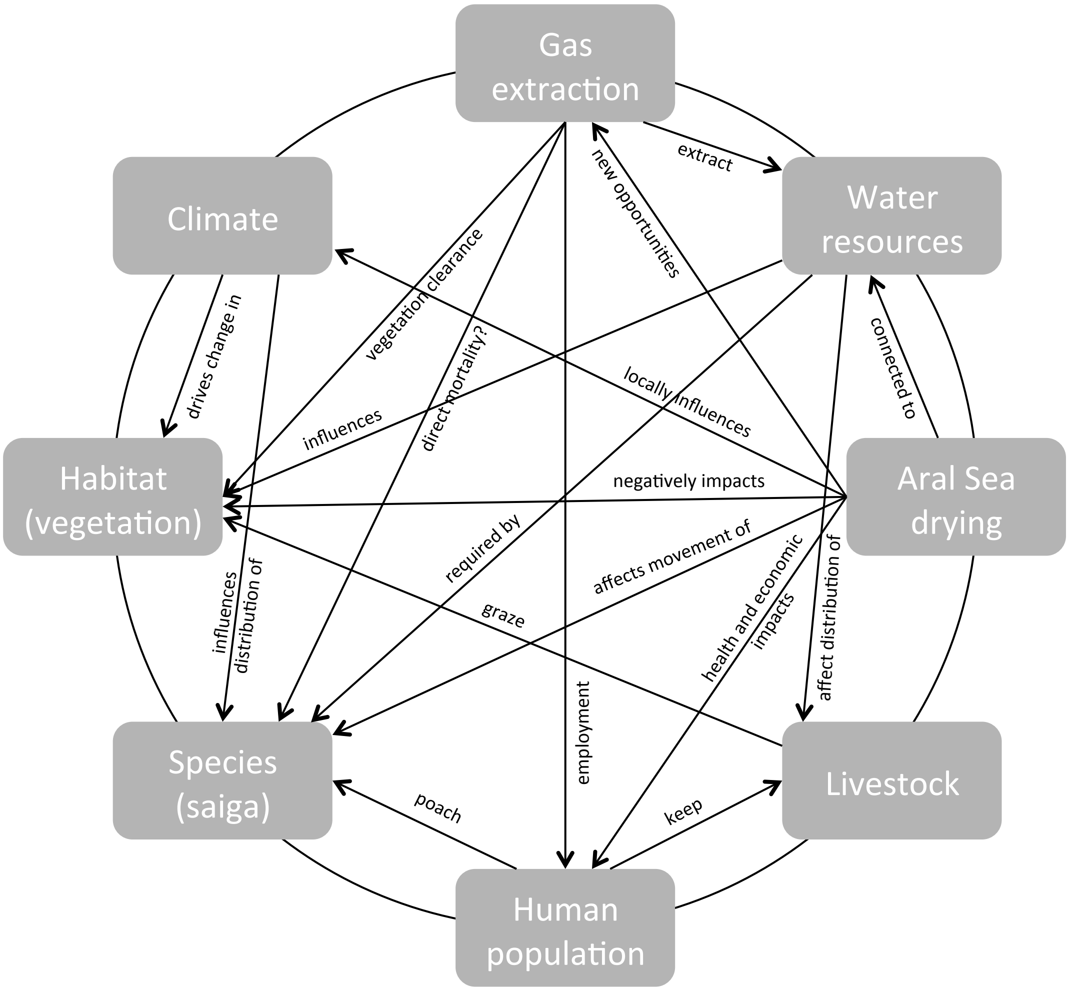
I outline what I consider to be important interactions between the main conservation targets and key factors, and then create a set of potential counterfactuals that consist of projections based upon the trends identified above. Finally, I consider the differences between these alternative counterfactuals, and use this to generate questions that remain to be asked for conservation intervention in the Ustyurt.

Table 5.1: Current baseline conditions for conservation targets, in the context of different drivers of change, alongside historical change during the last 100 years

Target	Driver	Current baseline	Trends over recent history
Habitat (vegetation cover)	Physical	• Largely intact semi-arid scrubland	• In some areas degraded by regional climate change and Aral Sea drying
	Social	• Cleared due to grazing near few scattered settlements	• Human population has grown
	Economic	• Cleared within small footprint of low density infrastructure	• Industrial infrastructure established during the last ~40 years • Livestock numbers have increased
	Institutional	• No private land ownership • Managed by state institutions	• State institutions emerged during the Soviet period
Species (Saiga antelope)	Physical	• Small population, mixture of migratory and semi-permanent	• Rapid decline from previously high abundance
	Social	• Heavily poached, linked to unemployment	• More intensive poaching since independence • Low but stable employment levels over the last decade
	Economic	• Uncertain impacts from industry and transport	• Infrastructure has caused some fragmentation of saiga range
	Institutional	• Managed by state institutions • Receive scant protection through existing protected area	• Experienced variable levels of protection and persecution

A number of interactions fundamental to conservation in the region involve the Aral Sea. It is important to note that, whilst the drying of the Aral Sea interacts negatively with social and ecological components of the system, it interacts positively with industry by enabling new exploration opportunities. Even in the absence of biodiversity offsets or similar mechanisms, industry is inseparable from conservation efforts – this is due not only to causing direct impacts upon ecological targets, but in terms of interactions with the human population, which in turn influences poaching and grazing (Fig. 5.6).

***Figure 5.6**: Potential interactions between factors key to conservation planning in the Uzbek Ustyurt. Interactions were hypothesized based on knowledge of the study area, and then tested using available data*



5.4.2 *Counterfactual*

Looking to the future, climate models reported by the IPCC indicate a warming trend in the region, with rises of 3.7 – 6.6 ^{0}C in mean temperature by 2099 (Meehl et al., 2007). This may have implications for vegetation growth. Further, given that the saiga migration itself depends partly upon temperature, being to an extent triggered by snowfall (Esipov et al., 2009), rising temperatures may eventually change migration patterns, especially if interacting with the barrier effect of the Kazakhstan border fence (e.g. rising temperatures and a barrier may encourage the migratory component of the Ustyurt saiga population to stay north, within Kazakhstan). With high interannual variability in snowfall, this influence would be hard to detect in the short term.

The national human population is growing, which is partially reflected in the regional population in Karakalpakstan (Table A2.3). The main employers on the plateau are the extractive and transport sectors (gas extraction and the railway, respectively), and the extractive sector is currently undergoing growth. As it grows, new or expanded settlements are likely to appear to service and provide labour for new industrial installations – with associated natural resource use expansion, especially water and pastures for grazing. In the long term, evidence suggests that inappropriate grazing regimes can lead to desertification (Savory, 1999), and there is visual evidence of this in the Ustyurt (Fig. A2.11). Localized desertification might therefore be expected to increase across the plateau as a result of expanding industry, overall population growth, and sustained or increasing numbers of livestock. Cotton will probably remain an important component of the economy for the foreseeable future. As a result of the requirements for both cotton irrigation and natural gas exploration on the dry seabed, a purposeful reversal of the drying of the Aral Sea on the shores immediately bordering the study area seems unlikely in the near future. Instead, drying through over-extraction of water resources may continue or accelerate.

Counterfactual for habitat-based offsets

In physical terms, whilst green vegetation cover appears rather stable across the landscape, it is likely that in the future it will increasingly be influenced by global climatic factors as the Ustyurt becomes warmer and more arid. Social and economic factors are likely to drive an increasing number of areas that feature highly localized vegetation clearance, as a result of livestock grazing and industry. The institutional factor with the greatest potential to influence cover is land tenure reform, although it is not currently thought that this will take place in the near future.

Based on this information, one counterfactual for the Ustyurt habitat could be a stable habitat characterized by small patches of intense clearing and fragmentation, but with some ongoing landscape scale deterioration in the longer term. Pertinently, it was found in Chapter 4 that the use of biodiversity offsetting could be most appropriate for ecosystems that are deteriorating. This suggests that biodiversity offset projects could be required to demonstrate NNL through restoring areas of reduced vegetation cover.

An alternative and feasible counterfactual would be one in which the changing climate caused increased biodiversity and vegetation cover by creating more tolerable winter temperatures, the human population did not grow further (as per the Muynak region, Table A2.3) and so resource use did not increase, and economic or institutional factors caused the drying of the Aral Sea to stop or even reverse (Table 5.2).

Counterfactual for species-based offsets

There is no reason to conclude that the causes of species loss in the Ustyurt over the last century are abating or will do so in the near future , i.e. hunting or persecution of fauna to the extent that they disappear or almost disappear, increasing barriers to migration through infrastructure and fencing, and implementation of irrigation practices that drain the Aral Sea with indirect implications for wildlife (Micklin, 2007). In the longer term i.e. over decades, climate change might influence the saiga migration, but otherwise physical factors are unlikely to be of concern. Social and economic factors, conversely, are crucial, and there is a constant threat of extirpation of this population. In the absence of any further conservation intervention, it is likely that the Uzbek Ustyurt population will become extinct over the next decade.

Since the *species* counterfactual, then, is likely to be one in which the saiga population will remain low or potentially disappear entirely, an offset scheme targeted at saigas would need to realise an increase in population to be additional (i.e. by removing poaching pressure). However, a combination of climate change and new infrastructure - the Kazakh border fence - may well conspire to prevent the migration of saigas into Uzbekistan at all. As such, another measure of the success of any offset scheme could be whether it *maintained* an open and demonstrably safe migratory route for the saigas into Uzbekistan.

Alternatively, the small resident population of saigas could become a focus for an offset scheme, aiming to maintain or increase this population but without the expectation of returning to the very large migratory population of the past. This would conceivably influence conservation planning and biodiversity offset opportunities, as, if the overall objective was to conserve saiga and maintaining migrations into Kazakhstan became untenable; e.g. as a result of the border fence, or climate change; then focus could instead be directed towards management of resident saiga populations *in situ* (Table 5.2).

Table 5.2: *Projected counterfactual on the scale of decades, for conservation targets based on analyses in this manuscript, in the context of different drivers of change. Includes an alternative counterfactual projection*

Target	Driver	Counterfactual	Alternative counterfactual
Habitat (vegetation cover)	Physical	• Stable in the near future • Some deterioration in the long term	• Reverse of Aral Sea drying • Increase in green vegetation cover
	Social	• Habitat cleared over an increasingly large area near to expanding settlements	• Settlements do not grow, additional habitat not cleared
	Economic	• Habitat cleared over an increasingly large area near to expanding industrial infrastructure	• Economic activity moves away from sectors with large habitat impacts e.g. cotton
	Institutional	• No fundamental change in institutional arrangements	• Biodiversity becomes key component of impact assessment • Land tenure reform results in more private ownership
Species (Saiga antelope)	Physical	• Short term, limited effects • In the long term, climate change may limit migration	• Small resident populations remain, migration no longer relevant
	Social	• Saigas poached to extirpation	• Incentives to poach decrease, saiga population stable
	Economic	• Industrial activities expand, with some associated saiga mortality	• Infrastructure increasingly divides up saiga range and limits movement
	Institutional	• No increase in protection • Political requirements result in maintained Kazakh border fence, increasing mortality and limiting movement	• Protection increases • Fence impact is mitigated, reducing barrier effect

The combination of Table 5.1, Table 5.2 and Figure 5.6 provides a basic frame of reference for conservation intervention, in terms of historical context, drivers, trends, interactions and projections. A key point, however, is that it is possible to develop alternative and often equally feasible frames of reference based upon analyses such as these (Table 5.2). Therefore, having developed two possible frames of reference, I now highlight specific questions that would allow the most likely counterfactual to be established.

5.4.3 *Outstanding questions*

Again, I structure the set of questions around a framework of physical, social, economic and institutional drivers (Table 5.3). From a physical point of view, the next stages in the ongoing drying of the Aral Sea will be important, as will monitoring how and if conservation targets respond to climate change. Changes in the human population in the Ustyurt will affect natural resource use, and a good case could be made for both a projected increase and a decrease in population. Economic activity is very likely to keep expanding, but the form this will take and the implications for conservation targets are not yet entirely clear. Finally, whilst institutional arrangements do not appear likely to change in the immediate future, there are a number of potential changes (such as land tenure reform and improved conservation capacity) that, if they occurred, would have important ramifications for conservation in the Ustyurt.

***Table 5.3**: Outstanding questions relevant to establishing the projected counterfactual, and associated management implications*

Factor	**Question**	**Management implication**
Physical	Will climate change limit and eventually stop the saiga migration into Uzbekistan?	Whether intervention focuses on protecting the migration route, or protecting resident Uzbek saigas
	Will overall vegetation cover decrease, remain stable or increase as a result of increasing temperatures?	Whether habitat maintenance or active restoration is needed to demonstrate NNL
	Will the Uzbek Aral Sea drying be halted, or reversed (as has been demonstrated achieved for the small northern Aral)?	Whether some proportion of conservation effort should go towards reduction of impacts from the loss of the Aral sea
Social	Will the Ustyurt human population grow or decline?	Influences various components of the counterfactual, as a result of the connection with intensity of natural resource use
	Will income and living standards improve?	May influence saiga poaching and who is carrying it out, and therefore appropriate conservation interventions
Economic	Will industry increase local employment rates and living standards?	May have an influence on living standards, natural resource use, livestock ownership
	Does industrial activity have direct and significant impacts upon wildlife?	Whether expanding industry will displace wildlife entirely, or merely fragment habitat
Institutional	Will land tenure reform take place over the coming decades?	How to implement biodiversity offsets as a robust policy option
	Will there be sufficient institutional capacity to ensure compliance with conservation legislation?	Whether biodiversity offsets are a feasibly robust policy option
	Will independent environmental observers eventually be encouraged back into the country?	Whether biodiversity offsets are a feasibly robust policy option

5.5 Discussion

This case study demonstrates, as might be expected, that the collation and analysis of data sufficient to accurately establish the baseline and projected counterfactual necessary for the frame of reference for a conservation intervention can be complicated and uncertain. The availability of robust data, and the ability to analyze and process these data and take account of validity of assumptions, is essential to developing a truly robust frame of reference for conservation interventions. However a reasonable picture can be built even on the basis of limited data, as I have done here – and this is certainly preferable to not developing a reference frame at all. On the basis of the analyses in this Chapter and the assessment which I describe in the Chapter to come (Chapter 6), I am unable to develop a defensible quantitative counterfactual e.g. a projected curve or distribution map for vegetation cover, or a population trajectory for the saiga antelope. Instead, the disparate numerical and qualitative analyses have to be drawn together in a qualitative way (Tables 5.2, 5.3).

Interactions between drivers of change can be as important as direct drivers themselves. For instance, the influence of ongoing climate change on saigas could be more important for vegetation than the direct impacts of climate change. Further, different forms of interaction could potentially lead to the same driver (e.g. industry) having conflicting impacts on conservation targets (e.g. both promoting poaching and disturbance, and providing alternative employment for current poachers); which must therefore be addressed differently. The framework I have used here, of categorizing drivers into physical, social, economic and institutional domains and considering their interactions, provides a useful way of breaking down the system and understanding its dynamics within a wider context.

In developing the frame of reference, I have attempted to partially address the common problem in doing so that assumptions are either not made explicit or are demonstrably wrong (Maron et al., 2013). The approach I have taken should make clear by implication where my key assumptions are. Further, by then developing more than one counterfactual, I have highlighted where my assumptions may be wrong (Table 5.2). The validity or otherwise of these assumptions can only be tested through ongoing monitoring and experimentation, which suggests the need to take an adaptive management approach to conservation when evaluating against a projected frame of reference.

Adaptive management provides a means for management under uncertainty, and involves the development of hypotheses to be tested in practice, which then inform future iterations of the management plan. It is conceptually popular within conservation science, but there are limited examples of it being used effectively in practice (Gregory et al., 2006; Armstrong et al., 2007). The set of questions I develop here (Table 5.3) could be framed as hypotheses, and thus usefully provides the basis for a form of adaptive management approach. It is noted that this Ustyurt case study does not meet a number of criteria suggested for appropriately implementing adaptive management – particularly the restricted spatial and temporal scale required for management, and the necessary institutional support (Gregory et al., 2006). But elements of the adaptive management approach, particularly the development of hypotheses at the outset that are then monitored for validity, seem highly relevant to the effective development of frames of reference.

In conclusion, I consider that understanding historical context, drivers and interactions are so important to the design of conservation interventions that a lack of data is an insufficient reason not to develop some form of reference frame. The approach I take here – of considering conservation targets in light of physical, social, economic and institutional factors – is useful in building a frame of reference. Further, it provides a means for making assumptions explicit, and leaving them open to further critical evaluation. Finally, by developing alternative feasible frames of reference, it is possible to outline testable hypotheses that can be used to improve future iterations of management plans, in a process that shares similarities with the approach of adaptive management.

Chapter 6

Quantifying habitat impacts of natural gas infrastructure to facilitate biodiversity offsetting

"And outside, the silent wilderness surrounding this cleared speck on the earth struck me as something great and invincible, like evil or truth, waiting patiently for the passing away of this fantastic invasion."

Joseph Conrad (from "Heart of Darkness", 1899)

"Nothing feebler than a man does the earth raise up, of all the things which breathe and move on the earth, for he believes that he will never suffer evil in the future."

Homer (from "The Odyssey", 8th Century BC)

6.1 Introduction

Land use change is a key driver of biodiversity decline through impacts on habitat availability (Mace et al., 2005). The extractive hydrocarbon, metal and mineral industries are among the most locally damaging forms of anthropogenic disturbance in many areas (Baillie et al., 2004). The negative impacts of extractive activities could be at least partially compensated for through biodiversity offsets (Chapter 2). However, formal and comprehensive quantification of the landscape impacts of infrastructure in the relevant environment will be essential if the potential of offsetting is to be realised (Quintero & Mathur, 2011). Accurate quantification of the residual impacts from development, which biodiversity offsets aim to compensate for, is often overlooked in the offset literature. Semi-arid habitats may be particularly vulnerable to disturbance from industrial activities (Lovich & Bainbridge, 1999), recovering poorly if at all (Fiori & Martin, 2003) and with full suites of constituent species persisting only in remnant undisturbed patches (Rapport & Whitford, 1999). Given that dryland (i.e. arid and semi-arid) biomes cover approximately 41% of the world's land surface and support over 38% of the global human population (Millennium Ecosystem Assessment, 2005), it is crucial that responses to industrial disturbance in these habitats are better understood and managed.

In semi-arid environments, habitats heavily disturbed by anthropogenic activities often have lower species richness compared to undisturbed areas (Simmers & Galatowitsch, 2010). Even low intensity and small-scale disturbances can have immediate and persistent effects (Forbes et al., 2001), with slow recovery times (Cui et al., 2009). The impacts of infrastructure upon vegetation in a variety of habitats – and of roads in particular, which are ubiquitous in regions of industrial extractive activity – have been well studied (Coffin, 2007; Trombulak & Frissell, 2000; Forman & Alexander, 1998). Irrespective of the sector that uses them, roads can impact ecosystems in numerous ways: via non-native species brought in by vehicles (Gelbard & Belnap, 2003); nitrous oxide and other pollutants produced by vehicles (Gadsdon & Power, 2009); and by subdividing populations and forming physical barriers to dispersal that alter demographics (Forman & Alexander, 1998). Thus species richness (Lee et al., 2012) and total vegetation cover (Fiori & Martin, 2003) may decrease in response to road construction. The magnitude of these effects will vary with distance from the actual disturbance. For example, in California, Gelbard & Harrison (2003) found that plant cover was significantly lower within 10 m of roads, and that native species richness was impacted even 1 km away. Conversely, roads may actually enhance overall species richness (Zeng et al., 2011), for example if associated disturbances provide favourable microsites for vegetation establishment (Brown & Schoknecht, 2001; Boeken & Shachak, 1994).

Natural resource extraction has had well documented impacts on wildlife as well as vegetation (E&P Forum/UNEP, 1997; Epstein & Selber, 2002; OGP/IPIECA, 2011; Kumpula et al., 2011). Habitat disturbance from industrial infrastructure can affect wildlife both spatially and temporally, for instance altering breeding patterns of birds (Walker et al., 2007) and grazing patterns of herbivores, leading to increased usage and pressure on surrounding undisturbed habitats (Vistnes & Nellemann, 2007). Noise associated with transport and other activities along infrastructure may disrupt the use of habitat by faunal species (Rabanal et al., 2010), and the physical obstruction caused by pipelines can alter animal movement across landscapes (Dyer et al., 2002; Curatolo & Murphy, 1986). In this manner, the spatial impacts of disturbance on the mean species abundance of vertebrate mammals in a range of habitats has been assessed to extend up to 5 km from infrastructure (Benítez-López et al., 2010). Vegetation responses may likewise be important to wildlife, altering resource availability and habitat structure. Further indirect effects may come from increased human use of the area that follows development, including water extraction, natural resource use, and hunting (Thibault & Blaney, 2003). Finally, as with roads, industrial activity can be associated with the spread of invasive alien species (E&P Forum/UNEP, 1997).

Many of the ecological impacts from industry can in principle be mitigated through biodiversity offsetting ('offsets'), but I have explained how offsets are beset with implementation challenges, including the difficulty of fully quantifying biodiversity losses (Chapter 2). Quantification of the scale of disturbance impacts for offset projects often focuses on the impacts of development sites or 'hubs', whereas the disturbance from the construction and operation of linear infrastructure such as roads that link 'hubs' is only recently being treated as something that could also be compensated for through offset mechanisms (Quintero & Mathur, 2011). Further, if linear infrastructure has a comparable impact by area to hub infrastructure across a landscape, which to my knowledge has not before been explored for offsets, then the scale of offsets required to achieve 'no net loss' of biodiversity could be greater than currently thought.

6.1.1 *Biodiversity and natural gas extraction in Uzbekistan*

Central Asian countries are predominately semi-arid and constitute an important proportion of the global semi-arid biome, 16% of which is found in Asia. In Uzbekistan, over 99% of the country by area is arid and semi-arid (White & Nackoney, 2003). Uzbekistan contains an estimated 27,000 species (USAID, 2001; UNDP, 2010a). The transboundary Ustyurt Plateau, which Uzbekistan shares with Kazakhstan, covers ~ 100,000 km^2 of Uzbekistan and contains 271 recorded vascular plant species (Gintzburger et al., 2011), several of which are on the IUCN Red List (Esipov & Shomurodov, 2011). As discussed in Chapters 3 and 5, the Ustyurt is also home to vertebrate species of high conservation priority, most notably the Critically Endangered saiga antelope *Saiga tatarica* (Mallon, 2008).

The Ustyurt is not only valuable from an ecological perspective, but also for the subterranean resources it contains. The natural gas industry in Uzbekistan is expanding (Dorian, 2006; Chapter 5). Development of the Ustyurt for hydrocarbon production since the Soviet era has resulted in extensive infrastructure growth, including exploration and extraction sites, pipelines, and a substantial network of roads (UNDP, 2010a). Previously, there had been no quantitative evaluation of the impact of roads, or other components of natural gas infrastructure (e.g. pipelines, extractive facilities), upon biodiversity on the Ustyurt. Such an evaluation was deemed fundamental to the development of the frame of reference for the Ustyurt (cf Chapter 5), and the calculation of potential offset requirements (Chapter 7).

6.1.2 *Quantifying disturbance on the Ustyurt Plateau*

As discussed in Chapter 3, there are plans to design an offset policy in the Uzbekistan Ustyurt, to compensate for the ecological losses from development and conserve the regional saiga population, as part of a broader biodiversity conservation initiative for the Uzbek oil and gas sector (Chapter 3; UNDP, 2010a). Offset projects on the Ustyurt could be improved by the provision of sound scientific knowledge of the impacts of natural gas activity at the landscape level, including whether linear infrastructure impacts are important enough to necessitate inclusion in offset calculations. The types of impact I explore here - direct impacts on vegetation from industrial infrastructure, in an area of general conservation concern but without legal protection - could be considered to be unavoidable, residual impacts of the natural gas industry. These direct impacts are therefore appropriate for biodiversity offsetting (Quintero & Mathur, 2011), which is an idea at the forefront of considerations for conservation in the Ustyurt (Chapter 3). Condition and area-based assessments of vegetation losses and gains form the basis of a number of existing offset policies (Quétier & Lavorel, 2011), hence my focus on measuring impacts on vegetation quality (condition) and cover (area). I aim not only to quantify the local impacts of infrastructure on Ustyurt habitat, but also the landscape-scale habitat footprint of the oil and gas sector in a way that is relevant for the development of an appropriate biodiversity offset policy.

On the Ustyurt, the vast majority of roads created and used by the oil and gas industry are unpaved and formed from repeated vehicle travel rather than formal clearing and construction. Most roads are therefore temporary or seasonal, although arterial routes between established extraction facilities are large and effectively permanent. In addition to damage from direct impaction and clearing by vehicles, dust deposition could further impact plateau vegetation, as has been suggested by observations of dust cloud movement and settling following vehicle passage (Gintzburger et al., 2011). Dust can affect key physiological processes such as photosynthesis, respiration and transpiration (Farmer, 1993). In arid environments dust can coat leaf surfaces, altering their radiation balance (Grantz et al., 2003), and can alter nutrient cycling through effects on soil bacteria and fungi, potentially costly in a nutrient-limited environment (Forbes et al., 2001). Dust in this region may be particularly harmful to organisms due to the very high content of pesticide residues and heavy metals that have resulted from drying of the Aral Sea (Micklin, 2007). Strong winds are characteristic of this continental landmass, and so the dominant wind direction and subsequent dust deposition from disturbance sources have been suggested as potential drivers of vegetation response to disturbance on the Ustyurt (Micklin, 2007).

In this Chapter, I quantify habitat impacts of oil and gas extraction activities by first investigating the spatial extent of localised disturbance from infrastructure, measured in both plant species richness and vegetation cover. These two metrics together provide a broad measure of habitat condition in this region (Opp, 2005), have successfully been used in previous studies investigating the impacts of disturbance (e.g. Lee et al., 2012; Simmers & Galatowitsch, 2010; Fiori & Martin, 2003; Gelbard & Harrison, 2003), and provide a baseline for understanding the state of plant communities at a range of scales. As part of a team, I conducted field surveys throughout the region to determine whether: (1) there is a reduction in vegetation species richness and ground cover in disturbed relative to undisturbed (control) sites; (2) species richness and vegetation cover increase with distance from disturbance and reach background levels within 500 m from the disturbance; and (3) effects of disturbance persist over greater distances in areas downwind of the disturbance, according to dominant wind direction.

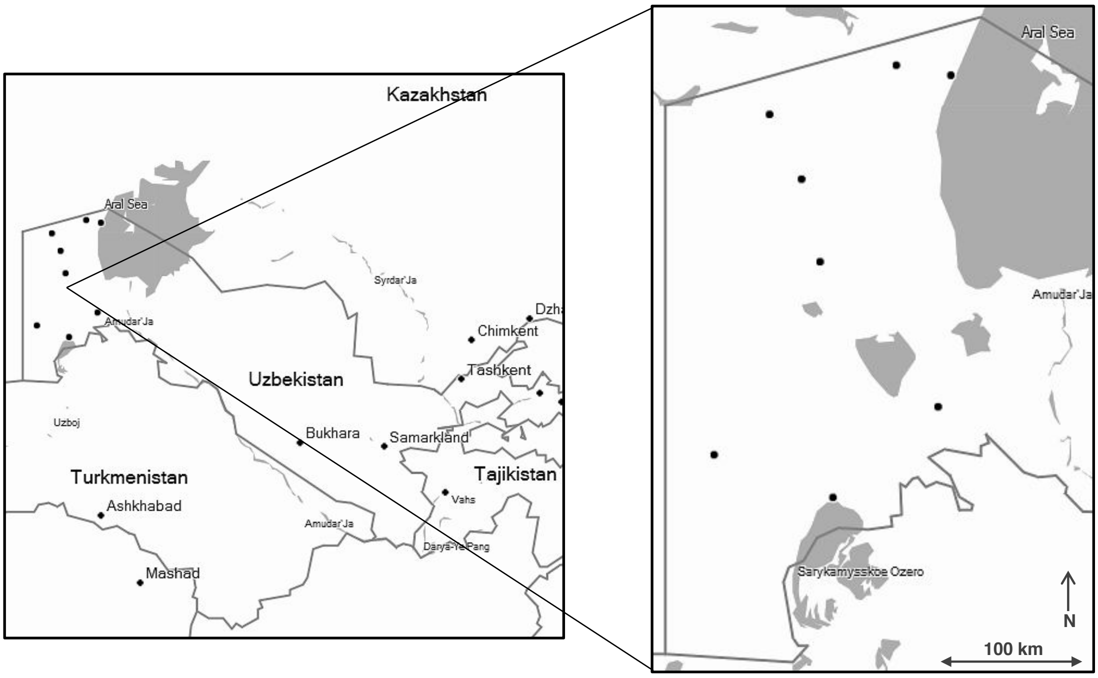
I use the results from field surveys to estimate the spatial extent of the local disturbance footprint of natural gas infrastructure across the entire plateau. By plotting the plateau-scale network of gas infrastructure, I extrapolate from localised measurements on the ground to the magnitude of the existing natural gas disturbance footprint in its entirety. This provides a measure of impact for vegetation that can inform ongoing development of a biodiversity offset policy for the region.

6.2 Methods

6.2.1 *Vegetation impact transects*

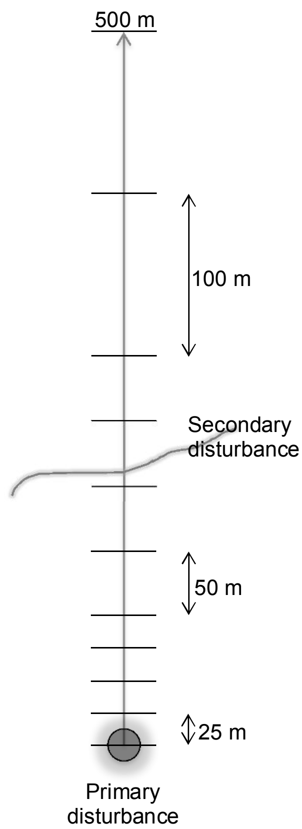
Following a pilot expedition in September and October 2011 to test sampling protocols and arrange access, an 18-day field expedition to the Ustyurt Plateau, Uzbekistan in May and June 2012, led by the United Nations Development Programme (UNDP), was undertaken for formal data collection. Transects employing the line intercept method (Canfield, 1941) were used to gather data on plant species richness (total number of species) and vegetation cover. Surveys were carried out at randomly selected locations within eight regions across the Ustyurt Plateau. The eight survey areas were selected by local experts to reflect the heterogeneous nature of Ustyurt biodiversity (Fig 6.1; Esipov & Shomurodov, 2011). Within each designated survey area, we completed as many transects as possible depending on time available, ensuring that they were spaced > 500 m apart. Transects originated at the centre of components of oil and gas infrastructure, and included extraction sites, pipelines, and unpaved roads known to be primarily used by oil and gas companies. For the purposes of this study, these major sources of disturbance were deemed ‘primary disturbances’. Transects were also undertaken at control sites without any of the above three types of infrastructure present for a minimum of 2 km. In total, 24 transects were completed: 17 disturbed sites and 7 control sites.

***Figure 6.1**: Uzbekistan, and the Ustyurt plateau. Circles on the detailed map are survey locations, with transects roughly equally distributed across the 8 locations. Maps created using Garmin 'BaseCamp'™ software.*



Disturbances were designated as either linear (road or pipeline) or point ('hubs' e.g. gas extraction sites). In each case, transects ran from the centre of the disturbance outward for 500 m. These were perpendicular to the disturbance for linear sources and at a randomly directed angle for hubs. The size and type of each primary disturbance was recorded, and then transects were arrayed in a 'spine and rib' formation, with increased sampling effort close to the disturbance source to detect any fine-scale vegetation responses (Fig. 6.2; Angold, 1997; Lee et al., 2012). The line intercept measurement method was applied over the 20 m length of each rib. To apply the line intercept method, we laid down a 20 m length of rope marked with 5 cm intervals. For each plant overlaid by the rope, we recorded the length of rope intercepted by the plant when viewed from directly above. The number of different species of plant was also noted, meaning that for each rib we could calculate both total vegetation percentage intercept (a proxy for percent cover) and species richness – species richness and percentage cover were the response variables used in analyses.

***Figure 6.2**: Design of spine and rib transects: the main 500 m spine transect originates from the centre of disturbance with 20 m rib transects bisecting it at set intervals: 25 m intervals between 0-100 m (where 0 m is the centre of disturbance), 50 m intervals between 100-300 m, and 100 m intervals between 300-500 m, giving increased sampling effort closer to the disturbance. The line intercept method was used to collect species richness and vegetation cover data along each rib. Secondary disturbances (i.e. small tracks) were also recorded.*



There is a network of small (< 3 m width) unpaved tracks across the Ustyurt, not primarily affiliated with the extractive industry, that spanned control and disturbed transects; these were labeled as secondary disturbances and noted and measured in surveys. Binary data on the presence/absence and width of secondary disturbances within 10 m and 50 m of ribs was recorded.

Due to the limited size of the dataset, the effects of each disturbance type could not be tested – roads, pipelines, and extraction sites – separately, but it was noted that patterns of vegetation impact and recovery appeared visually similar for each disturbance type.

6.2.2 *Methodological considerations and compromises*

It was important to ensure that sites were representative of the Ustyurt landscape as a whole, to allow scalable inferences about the impacts of infrastructure. Habitat on the Ustyurt is generally classified by local botanists into different key species associations, for example the widespread *Anabasis sp. – Artemisia sp. – Salsola sp.* association. Nevertheless, different associations are comparable in terms of both percentage cover and species diversity (Esipov & Shomurodov, 2011; Gintzburger et al., 2003). Surveys were thus completed throughout the Uzbek Ustyurt and within a range of association types, but different association types were not separated in the analyses reported here.

Survey areas were chosen such that, in general, each transect remained within one association type. In order to minimize observer site selection bias as far as possible for linear disturbances, the start point was selected by walking in a fixed direction along the road/pipeline for a pre-agreed amount of time from the point at which the survey vehicle stopped (5 minutes), or 500m from the end of the previous transect if more than one was being completed in a sequence. The point reached after this time was the transect start point.

6.2.3 *Statistical analysis*

All statistical analyses were conducting using linear mixed effects models to allow inclusion of both fixed and random effects (Bolker et al., 2009) in the 'R' statistical package (R Development Core Team, 2012). All models had species richness or vegetation cover as response variables. As species richness was in the form of count data, a Poisson error structure was specified for analyses. For percentage cover, data were arcsine square root transformed so as to follow a Gaussian rather than a Binomial distribution in analyses. Maximal models included the variables site type (control vs. disturbed), distance from disturbance, disturbance width (either <3 m or >10 m), presence and width of secondary disturbances, and transect direction (as a proxy for wind direction). Wind direction data were obtained via I. Aslanov (unpublished data) for 2009/10 from the hydro-meteorological stations in Jaslyk (central Ustyurt) and in Muynak (eastern Ustyurt). As data from both sites clearly showed dominant easterly and northerly wind direction, and local wind data were not available for the transect sites visited, I assumed that this was the dominant wind direction throughout.

The transect design resulted in 'ribs' being pseudoreplicated and nested within 'transect'. To account for this, 'transect' was fitted as a random effect within linear mixed effects models. Maximal models including all potential explanatory variables were simplified through stepwise deletion of highest order non-significant terms, and model comparison using ANOVA (Crawley, 2007). Models used in the analyses were:

```
lmer(species richness ~ control or disturbed sites * distance from
disturbance + disturbance width <3 or >10 m + (1| transect), family =
poisson);
```

and, after arcsine square root transformation of percentage cover data:

`lmer(percentage cover ~ control or disturbed * distance from disturbance + (1|transect).`

P-values for general linear mixed effects models, involving percentage cover as the response variable, were produced using Markov Chain Monte Carlo (MCMC) sampling with 10,000 iterations, following Bolker et al. (2009). The package `languageR` and function `pvals.fnc` (Baayen, 2011) were used with the `lme4` package (Bates et al., 2012) to run MCMC and obtain p-values for analyses involving percentage cover data.

6.2.4 *Landscape footprint of oil and gas activities*

Detailed official records for the spatial extent of oil and gas infrastructure on the Ustyurt were not available. Therefore I used two methods to map the spatial distribution of key oil and gas infrastructure. First, during two field expeditions to the Ustyurt (the 2012 main expedition and the 2011 pilot expedition), a GPS track log was kept of all oil and gas infrastructure (tracks, pipelines, roads, railways, compressor stations, wells) encountered throughout the Plateau. These data were used to create an infrastructure map, extrapolated as necessary to join up known pipeline routes, and with the addition of the coordinates of additional known gas extraction facilities (UNDP, 2010a).

Secondly, satellite imagery was used to map all linear infrastructure on the Ustyurt Plateau clearly visible at 70 km altitude (Google Earth, 2012). At this altitude, infrastructure known to be present from field observations was clearly visible, validating the method; it was assumed that other infrastructure similarly visible in the images, but not visited during the fieldwork, was also attributable to natural gas activity given the lack of other major infrastructural activities in the area. Lower altitude images, although more detailed, would have included less distinct infrastructure relating to general use of the area by herders or local traffic. It is noted that this method was not also used to map hub infrastructure: it was found difficult to distinguish with any confidence between hub gas extraction sites and other unrelated development hubs, such as settlements, takyrs (salt pans), and even water wells.

Hub disturbances consisted of the six major known gas extraction facilities: two which were visited during the fieldwork (*Shakpakty* and *Aqsholaq*) and four unnamed additional sites mapped by the UNDP (2010a) but inaccessible for visitation. Limited information was available on the spatial footprint of gas extraction facilities, and those that the expedition team visited in the field could not be directly surveyed. However, a recent Environmental Impact Assessment completed for the proposed Surgil gas development in Kazakhstan provides detailed area calculations for a major gas facility on the Ustyurt (Mott-Macdonald, 2012). These calculations were taken as reflective of the spatial footprint of a standard major gas facility in the region, and were used as representative data to calculate the footprints of the six hub sites. The major gas compressor plants at Jaslyk, Karakalpakia and Kubla-na-Ustyurt were excluded, because although they represent major infrastructure, they are based within substantial settlements that would have existed in any case. Therefore their footprint could not be attributed directly to industrial gas activities.

The physical footprint (in km^2) occupied by each infrastructure type was calculated separately for both the field-based and satellite-based mapping techniques. In both cases I calculated the footprint using mapped lengths with my field-measured widths of infrastructure components to obtain the overall area. The outcomes of these two different mapping approaches were compared qualitatively. Averaging the footprints obtained from both approaches produced an estimated total footprint of linear infrastructure, and the difference was treated as a simplistic measure of uncertainty. This was then summed with the total hub infrastructure footprint, to give an estimated total footprint of oil and gas-related infrastructure on the Ustyurt.

6.3 Results

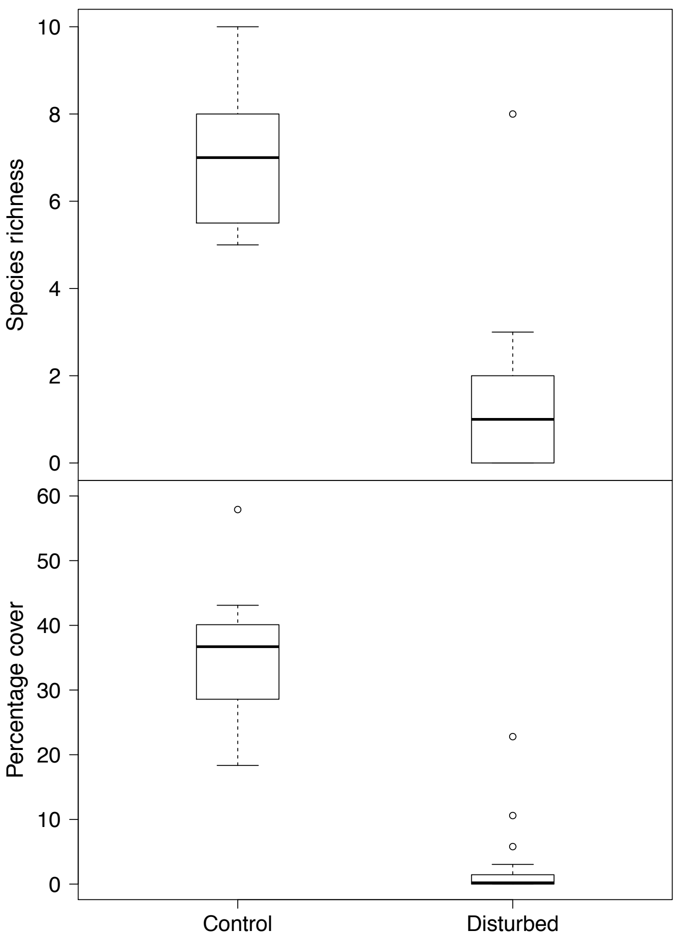
6.3.1 *Vegetation impacts of infrastructure*

Oil and gas associated disturbance had a negative effect on species richness ($z = -6.2$, $P < 0.001$) and vegetation cover ($t = -4.7$, $P < 0.001$) compared to control sites (Table 6.1; Fig. 6.3).

Table 6.1*: Overview of vegetation response to primary disturbances. The distance at which vegetation is significantly different to baseline levels (those at the 500 m sampling point) is shown. Number of transect sites = 17, number of control sites = 7*

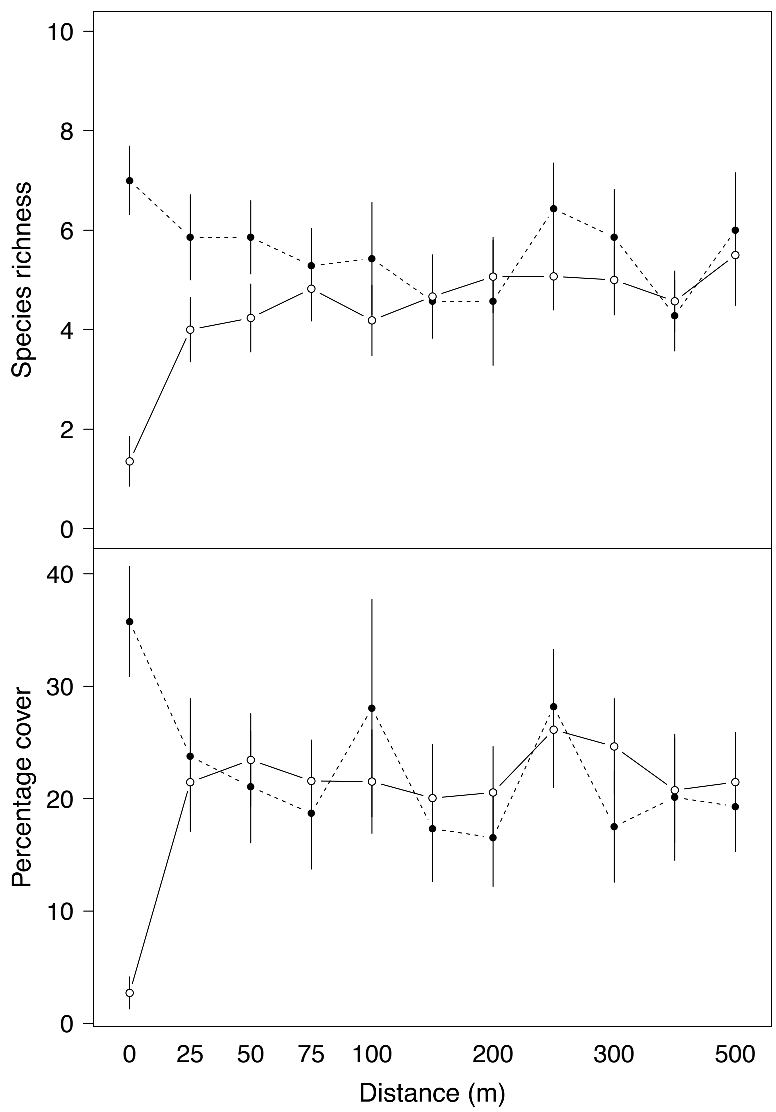
Effect of primary disturbance		**Distance at which vegetation is significantly different to baseline levels**			
Species richness	**Percentage cover**	**Species richness**		**Percentage cover**	
Response	Response	Distance (m)	Response	Distance (m)	Response
z = -6.17 *p* = 6.87e-10 ***	*t* = -4.68, *p* = 0.0000 ***	0	*z* = -4.66 *p* = 3.14e-6 ***	0	*t* = -5.2 *p* = 0.0000 ***

Figure 6.3*: Boxplots showing differences in vegetation responses at the 0 m 'rib' transect of control and disturbed sites. Differences in both species richness (z = -6.2, p < 0.001) and cover (t = -4.7, p < 0.001) are significant.*



Outside of actual infrastructure, species richness and cover increased to baseline levels exhibited by control sites (Fig. 6.4). Species richness and cover were only significantly different to baseline levels at the site of disturbance itself (0 m); at 25-500 m from disturbance, species richness and cover were not significantly different from baseline levels. The sampling design did not anticipate complete attenuation of disturbance-associated vegetation differences within 25 m of infrastructure, and was therefore not set up specifically to test for patterns between 0 and 25 m distance.

Figure 6.4*: Interaction plots for richness and cover with distance. Hollow points represent disturbed sites, solid points represent controls. Graphs produced using "Sciplot" with 95 % confidence intervals displayed.*

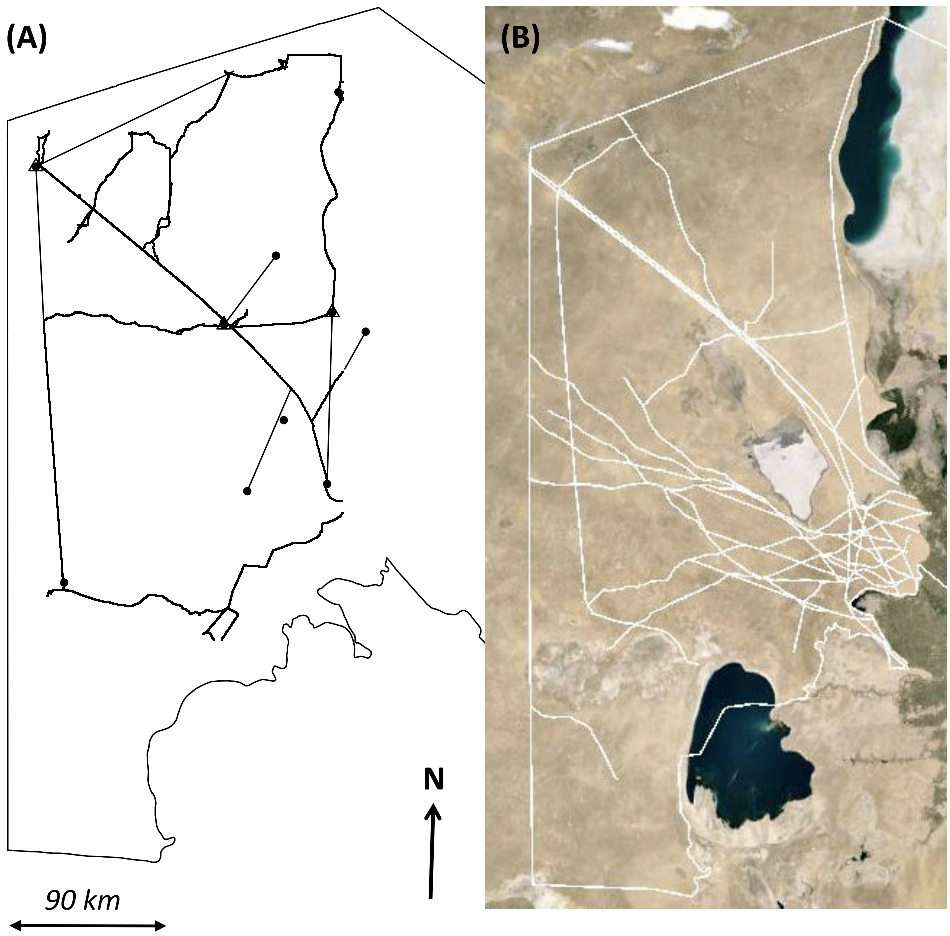


There were no significant directional impacts upon either richness or cover, as would be expected if wind-deposition of dust or exhaust pollutants was an important factor for vegetation. Species richness was higher for small (< 3 m width) and large (> 10 m width) linear disturbances than it was for medium (3 – 10m) linear disturbances ($z = 2.4$, $P < 0.05$ and $z = 2.8$, $P < 0.01$ respectively).

6.3.2 *Footprint of infrastructure at landscape level*

The two mapping approaches gave visually comparable estimates of linear infrastructure presence (Fig. 6.5). In the northeast of the plateau, infrastructure that was missed using Google Earth was recorded using GPS on the ground. This casts some doubt on the ability of the remotely sensed approach to capture all key components of infrastructure in an investigation of this type. Conversely however, in the southeast of the plateau, the Google Earth map shows linear infrastructure that was not surveyed during the expedition. Although this is simply because it was not possible to cover the plateau in its entirety during the expedition, the potential value of using the Google Earth approach to remotely map infrastructure is clear. Whilst neither approach is comprehensive, the two arguably complement each other. The field-collected data suggested a current linear infrastructure footprint = 100 km^2; the Google Earth method suggested a footprint = 63 km^2.

Figure 6.5: *Mapping of spatial extent of gas infrastructure. (a) mapped using GPS data, where lines = roads, tracks and pipelines, circles = settlements with gas infrastructure, triangles = known facilities used or created by the gas industry. Black lines = mapped using GPS data, grey lines = known infrastructure not mapped using GPS data; (b) linear infrastructure mapped using Google Earth (2012). White lines represent linear infrastructure – roads, railways, pipelines – associated with the oil and gas industry. Note that the large white region near the centre of this figure corresponds to Chimboy lake (Chapter 5).*



It was not possible to survey hub extraction facilities for health and safety reasons, but it could be seen from visual inspection that vegetation was cleared in a similarly comprehensive manner within hub infrastructure as it was for linear infrastructure. With an assumption of immediate attenuation of disturbance effects, as conservatively documented for linear infrastructure, the footprint of extraction sites on vegetation can be calculated as the actual ground surface area occupied by extraction facilities. Extrapolating from the 23 km^2 proposed Surgil facility (based on data provided in Mott-Macdonald 2012; Table 6.2), yields an estimate of a 138 km^2 footprint.

Table 6.2: *Breakdown of estimated contributions to total footprint for proposed Surgil Gas Extraction Facility (based on data in Mott Macdonald, 2012). Assumed dimensions and calculations by the authors are given in italics in the 'Area' column. Assumed road widths, extraction radii, pipeline widths and railway widths were based on filed observations. Values in regular font come from the report itself. Connecting ('linear') infrastructure on site is included in the site ('hub') footprint*

Component	**Sub component**	**Area (m^2)**
Main facilities		
Ustyurt Gas Chemical Complex	Main	970,100
	Storage	27,000
	Wastewater pond	240,000
Settlement	Area	850,000
	Road to settlement	5km *by 10m = 50,000*
	On-site camp	? *(treated as zero)*
Gas extraction	Wells	133 wells (*radius of 170m*) *= 12,075,340*
	Gas gathering stations	6 stations (*radius of 170m*) *= 544,752*
Complex Gas Treatment Unit	Expansion of existing area	? *(treated as zero)*
Water	Water treatment plant	? *(treated as zero)*
Connecting infrastructure		
Roads	Access (UGCC)	9ha = 90,000
	Gas gathering stations	6 roads, *3km by 25m = 450,000*
	Gas wells	133 wells, average distance 4km, *road width of 10m = 5,320,000*
Pipelines	Gas	115km *by 13m* = 1,495,000
	Gas sales	9km *by 13m* = 117,000
	Gas booster station	? *(treated as zero)*
	Water	27km *by 13m* = 351,000
Railway	Access	7km *by 30m* = 210,000
Electricity	Transmission line (road)	18km *by 10m* = 180,000
Total		
	Main facilities	14,757,192
	Connecting infrastructure	8,213,000
	SUM	**22,970,192**

Summing the estimates for linear and hub infrastructure, and using the difference in linear infrastructure estimates from the two different methods as a measure of uncertainty, suggests a total current footprint of 220 ± 19 km^2. Again, this is in the context of a plateau that covers ~ 100,000 km^2 of Uzbekistan. Under this rough estimate, linear infrastructure constitutes 37 ± 6 % of the total oil and gas infrastructure footprint.

6.4 Discussion

6.4.1 *Local impact of disturbance*

This study shows that natural gas infrastructure, both linear and hub, has a substantial effect on local species richness and vegetation cover in a semi-arid ecosystem. But, contrary to expectation, effects are effectively limited to the footprint of physical infrastructure itself, at least insofar as the sampling scheme was equipped to measure. Future sampling that focuses on the zone between 0 and 25 m distant from oil and gas infrastructure is key to deriving a more precise estimate. The results did not support the idea that dust deposition significantly impacts the local cover or richness of vegetation. Future work could further quantify dust deposition including its more subtle or widespread effects (Goossens & Rajot, 2008), and investigate whether directional effects on vegetation metrics vary spatially across the Ustyurt. In conjunction, these two findings suggest that the condition of vegetation was only reduced within the direct footprint of infrastructure, and there it was effectively reduced to zero: making the development of a condition-area type biodiversity offset metric, in this case, straightforward.

This Chapter does not assess all impacts associated with oil and gas development. For instance, the industry certainly abstracts a large amount of water, and may result in the keeping of additional livestock on the plateau, both of which may impact upon regional biodiversity; equally, other elements of infrastructure are thought to impact specific components of biodiversity, such as power lines causing bird mortality (Mott-Macdonald, 2012). However, it does quantify the limited effects of infrastructure on vegetation as one component of a potential biodiversity offset policy in the region. My work adds to current knowledge of disturbance effects from infrastructure on plant communities in climatically severe areas. These include Arctic tundra (Kemper & Macdonald, 2009), steppe environments (Fiori & Martin, 2003) and other semi-arid desert regions (Simmers & Galatowitsch, 2010). Oil and gas exploration is burgeoning in these habitat types; consequently, providing accurate assessments of the spatial scale of infrastructure disturbance is essential if national governments intend to ensure that any negative ecological impacts associated with the oil and gas industry are compensated for. But, as previous work strongly suggests (Benítez-López et al., 2010), the spatial scale of local disturbance is likely to extend beyond the direct physical footprint for infrastructure, when considering other components of the ecosystem such as faunal species. Hence the area calculated here is a minimum estimate. Taking other effects of infrastructure into account is also likely to increase the relative importance of linear infrastructure in calculating overall compensation requirements, because of the known disruptive effects of anthropogenic activity on wildlife (Benítez-López et al., 2010), particularly migratory species such as the saiga antelope (e.g. Singh et al., 2010b). Further work could include investigation of the extent to which disturbance effects can be seen in other taxonomic groups on the Ustyurt such as invertebrates, reptiles, mammals and birds, and whether the spatial scales of responses differ.

There were often observed to be relatively undisturbed 'humps' between tyre tracks, and there is the possibility that water pooling in tyre ruts may create favourable establishment sites in arid ecosystems (Briones et al., 1998; Brooks & Lair, 2005; Boeken & Shachak, 1994). Furthermore in some cases and especially for smaller, less busy roads, it is possible that roads had not been used for some time, allowing some minor colonisation by vegetation. This may explain why cover & richness were not always zero within the footprint of disturbances.

6.4.2 *Scaling up to landscape level*

The use of mapping tools in conjunction with local-scale quantification of direct infrastructure impacts allowed estimation of the disturbance footprint from infrastructure for the whole region. This estimate is based on a number of broad assumptions, and is therefore an order-of-magnitude estimate only. The two approaches used were complementary: mapping using GPS data ground-truths infrastructure sites, whilst mapping using Google Earth captures all infrastructure of a certain size. Furthermore, Google Earth images present a single snapshot of the Ustyurt (without clear dates for images in the version used: Google Earth 6.0, 2012), and so does not reveal the seasonality of some off-road routes. These mapping tools do however, when used in conjunction, provide a useful estimate of the total infrastructure footprint in this region. This is likely to be a conservative estimate due to my inability to map all infrastructure associated with oil and gas activity, or to account for indirect impacts from infrastructure presence such as those upon fauna like the saiga – the latter is a topic I return to in Chapter 7.

The direct footprint of the oil and gas sector is small compared to the area over which it is spread (~ 100,000 km^2), in fact constituting only approximately 0.2 % of the plateau by area, but this does not necessarily mean that impacts are negligible on the landscape scale. Measurement of the direct footprint does not account for the fact that infrastructure may well influence the ecology of a much wider area, for instance by changing vertebrate behaviour up to 5 km away from infrastructure (Benítez-López et al., 2010). The distribution of infrastructure is also important; as, beyond causing direct disturbance to wildlife, extensive linear infrastructure may physically fragment the landscape, which would particularly affect wide-ranging species (Rytwinski & Fahrig, 2012) such as saigas. Quantifying these impacts is beyond the scope of this thesis, but these considerations are clearly important in designing biodiversity offsets, and in conservation planning for the wider landscape.

6.4.3 *Designing biodiversity offsets*

Biodiversity offsetting has been proposed as one mechanism within a raft of initiatives for reducing negative ecological impacts associated with infrastructure development on the Ustyurt (UNDP, 2010a). Offsets might enable the restoration of highly degraded areas of Ustyurt habitat, commensurate with the land cleared for further infrastructure development (UNDP, 2010a; Chapter 3). A prerequisite for implementing offsets is the evaluation of biodiversity losses, for which some metric based on both condition and area of habitat impacted is often used (Quétier & Lavorel, 2011). My estimates of total disturbance at landscape-level give a basic condition-area metric that could be used in partially quantifying the vegetation offset requirements for the region. The estimate relates to development that predates any national offset policy, and sums habitat lost over several decades, so does not constitute actual offset requirements. Furthermore, counterfactual trends in the landscape; including the influence of other industries, human settlement and agriculture, and the influence of drying of the Aral Sea (Chapter 5); would need to be considered in the design of any offset policy for the plateau. However, my approach does outline a basic methodology for calculating vegetation loss in the region and contextualises the magnitude of potential additional losses. The losses calculated here are used as a basis for discussing potential biodiversity offset schemes for the Ustyurt, in the two Chapters to come next.

More generally, my study shows that offset projects should consider linear infrastructure as well as hub development sites themselves, as linear infrastructure can constitute a large proportion of the total area impacted by developments. Finally, this investigation highlights the assumptions that must be made in calculating even one type of ecological loss for biodiversity offset schemes. In turn, this supports careful consideration of uncertainty in the development of offset policies, and the need to set up mechanisms to account for this uncertainty such as multipliers (Moilanen et al., 2009) and conservation banking (Bekessy et al., 2010). In the development of offset policy, it can often be assumed that the uncertainty dealt with by multipliers and biodiversity banking is that associated with the offset action (the biodiversity gain from the offset), and not the quantification of the original impact (e.g. Defra, 2011). But Moilanen et al. (2009) and Bekessy et al. (2010) suggest instead that multipliers include the uncertainty around the damage incurred, i.e. the conservation value of land that is developed, and that biodiversity banking should recognize uncertainty in impacts as well as actions. Considering uncertainty in the quantification of residual development impacts on biodiversity, as I have begun to do for the study reported upon in this Chapter, is consequently crucial for the implementation of offset actions.

Chapter 7

Comparing biodiversity offset calculation methods with a common case study

"Once he tried to feed all the animals in all the world in one day, but when the food was ready an Animal came out of the deep sea and ate it up in three mouthfuls. Suleiman-bin-Daoud was very surprised and said, 'O Animal, who are you?' And the Animal said, 'O King, live for ever! I am the smallest of thirty thousand brothers, and my home is at the bottom of the sea. I heard that you were going to feed all the animals in all the world, and my brothers sent me to ask when dinner would be ready.'"

Rudyard Kipling (from "The Butterfly that Stamped', 1902)

"All animals are equal, but some animals are more equal than others."

George Orwell (from "Animal Farm", 1945)

7.1 Introduction

Since the essential objective of most offset policies is to achieve no net loss (NNL) of biodiversity, or some component of biodiversity, alongside economic development (BBOP, 2012; Chapter 2); then acceptable local losses at the sites of activity must be compensated for by producing equivalent biodiversity gains elsewhere. In the previous Chapter, I focused upon the calculation of losses in relation to biodiversity offsetting. A key challenge to effective offsetting, having quantified the biodiversity losses associated with development, is the calculation of the biodiversity gains required in order to deliver NNL (Quétier & Lavorel, 2012). Losses and gains are separated in space and time, and potentially differ in biodiversity type; hence there is a need for a common metric of ecological equivalence in order to compare them.

The reader will recall that the term ‘offset’ encompasses a range of approaches to comprehensive biodiversity compensation, from specific habitats to generalisable frameworks (Madsen et al., 2011; Doswald et al., 2012), sometimes conflating these with other approaches to ecological compensation. Several different methodologies exist for calculating the ecological gains required to compensate any given development project: some use area of habitat as a proxy for both losses and gains (e.g. US Wetland Banking – which is actually concerned with no net loss of wetlands by acreage and function, and is therefore not focused upon “biodiversity” offsetting *per se*); some use a combination of area and ‘condition’ or ‘functionality’ of the habitat (e.g. Canadian Fish Habitat); others combine area and condition and compare this against some benchmark ‘pristine’ state (e.g. Australian vegetation offsets); and some focus on species, calculating the area of habitat necessary to support a given population size (e.g. US Species Banking) (McKenney & Kiesecker, 2010; Quétier & Lavorel, 2012). More recent offset policies include the one currently being piloted in England, which uses a condition-area metric (Defra, 2011), and the policy in the Western Cape of South Africa, which incorporates explicit consideration of ecosystem services (Brownlie & Botha, 2009). Certain methodologies were developed for specific circumstances, such as the regulations governing clearing of native grasslands in the Australian state of Victoria; while others, such as wetland banking in the United States, are intended as general frameworks.

Despite the underlying NNL objective, it is not clear how such methodologies compare to one another, because there has been no study of variation in NNL achievement when applied to a common case study. Here, I attempt such a study, while providing a conceptual basis for exploring the extent to which different offset methodologies interpret and achieve NNL. This work can also provide insight into the degree to which national offset policies concur on the ecological requirements for NNL, contributing to debate as to whether international offset trades might be possible, e.g. trading impacts in one country for offsets in another.

7.1.1 *Objective of biodiversity offsets*

Whilst offsets ostensibly endeavor to achieve NNL of biodiversity overall, each approach to offsetting inevitably focuses upon specific sub-components of biodiversity as proxies for total biodiversity (Chapter 2), by which I mean “the sum total of all biotic variation from the level of genes to ecosystems” (Purvis & Hector, 2000). Biodiversity offsets are often (e.g. Quétier & Lavorel, 2012) grouped into habitat-based, species-based, or other calculation methods: i.e. whether offsets focus on vegetation grouped into habitat types, have an explicit focus on particular species (usually fauna), or consider alternatives such as ecosystem services. Here, I group a set of ecological compensation measures – not all of which are true biodiversity offsets, but which feature a NNL requirement – into those which are habitat-based or species-based. I do not consider ecosystem service based offsets here as, although progress has been made (e.g. South Africa; Brownlie & Botha, 2009), a full methodology for such offsets as yet to be developed.

Habitat-based approaches generally rely on some measure of both the area and ‘condition’ of habitat to calculate losses and gains (BBOP, 2012). Victorian native grassland compensation in Australia uses a ‘habitat hectares’ calculation method, based upon the method outlined by Parkes et al. (2003). Biodiversity losses and gains are compared to a ‘pristine’ reference state, and measured in hectares multiplied by ‘condition’, the latter based upon a set of criteria including vegetative recruitment and the presence of invasive weeds. A variant on this approach is being trialed in the UK (Defra, 2011). Alternatively, the US Wetland Banking calculation method calculates NNL based upon area of wetland lost or gained only (McKenney & Kiesecker, 2010), and is necessarily not a true ‘biodiversity’ offset as its focus is upon acreage and function of wetlands.

Species-based approaches tend to also use calculation methods based upon the spatial extent and quality of biodiversity losses or gains, but instead of ‘condition’ they rely upon some measure of the suitability of an area of habitat for the target species. Conservation Banking in the US takes this approach for a suite of protected species (US FWS, 2006), as does the EU for species protected under the Birds and Habitats Directives (McKenney & Kiesecker, 2010).

A critical additional consideration is that offset policies do not always restrict trades of biodiversity losses and gains to being ‘like for like’ or ‘in kind’. Whilst trading in kind is encouraged (BBOP, 2012), sometimes conservation objectives could be better served by trading ‘out of kind’ (Brownlie & Botha, 2009; Habib et al., 2013). For instance, some policies allow for trading of losses in low value conservation areas for gains in high value conservation areas (e.g. the UK policy; Defra, 2011) or even encourage it (e.g. the South African policy; Brownlie & Botha, 2009); or allow for trading losses in the habitat of one species for gains in that of another (e.g. US species banking).

7.1.2 *Testing methodological approaches against a common case study*

One industrial proponent of offsetting is the extractive sector (Quintero & Mathur, 2011), and as discussed offsets have been proposed as a means to compensate for the biodiversity impacts of the oil and gas (O&G) sector upon the Ustyurt plateau, Uzbekistan (Chapter 3). I use the Ustyurt as a common case study for comparing different methodological approaches, exploring the offset requirements that could have been imposed for O&G infrastructure developed over the last 40 years under a range of methodologies. The research is timely because of the regulatory framework for offsetting that Uzbekistan is currently developing (Chapters 3, 5), requiring a methodological framework for comparing gains and losses.

Habitat clearance and disturbance to threatened fauna have both been identified as important ecological impacts of the O&G industry in the Ustyurt (UNDP, 2010a; Chapter 5). Vegetation clearance due to O&G activity has been quantified (Chapter 6), allowing the application of habitat-based offset calculation methodologies to the case study. For species-based methodologies, the flagship species is the saiga. The main driver of saiga population decline is poaching (Kühl et al., 2009), which is not directly attributable to the O&G industry. Human presence and infrastructure have behavioral impacts upon saigas, modifying their use of habitat (Singh et al., 2010b; Salemgareev, 2013), but there are no data on these impacts for the Ustyurt. However, in order to provide a theoretical estimate of potential disturbance, I use estimates from a relevant meta-analysis study that addresses the influence of human disturbance upon mammals to develop a species-based calculation method (Benitéz-Lopez et al., 2010). Although the main aim of the study is to compare the extent to which different established offset methodologies might result in NNL of biodiversity, either habitat or species, it could also provide a basis for further research that evaluates which offset methodologies might be most appropriate for implementation in the case study region, the Uzbek Ustyurt.

7.2 Methodology

I considered the 'area' and 'condition' of habitat impacted by the O&G industry. Condition can vary between 0 and 1, with 0 representing habitat that is degraded to the extent that it has zero conservation value, and 1 representing a pristine site (i.e. the 'perfect' example of a given habitat type). In reality, a patch of habitat can never achieve a condition of 0 or 1, and would always fall somewhere in between. However, for the purposes of my analysis, to which this constraint is not material, I assume that condition scores between and including 0 – 1 are possible. For species-based methodologies, condition might represent the suitability of the habitat for mammals.

The current state of the plateau was considered to be a mosaic of degraded habitat containing infrastructure, patches of pristine habitat, and patches of habitat degraded by other influences. I assumed that any patch in the plateau has the potential to be raised to a condition value approaching 1 for both habitat as well as species, as part of an offset scheme. Whilst the ability to successfully restore habitat as part of offset schemes cannot be assumed in every case (Maron et al., 2012), the technology that enables the effective re-vegetation of semi-arid scrubland has existed for some time (e.g. Gintzburger & Skinner, 1985). Saiga antelopes have relatively broad habitat requirements and in the past have been found throughout the potential offset area. I calculated ecological losses based on the impact of known existing industrial infrastructure. These losses were then converted into offset requirements (i.e. necessary gains) using each of the regulatory offset policies included in the study (Table 7.1; see below). All impact calculations, for both habitat and species offsets were made using a novel approach of developing a 'functional form of disturbance' caused by infrastructure.

Table 7.1*: Regulatory biodiversity compensation policies with a NNL objective which were evaluated in this study. Adapted from the list in McKenney & Kiesecker (2010), with the addition of the UK policy, which is currently under evaluation.*

Compensation policy	**Calculation method**	**Target**	**Reference**
1. US (wetland banking)	Area of wetland lost, or length of waterway lost	Habitat	US ACE et al., 1995
2. Australia (Victorian native grassland compensation)	A compound calculation method ('habitat hectares'); a combination of area and 'condition' of the habitat lost compared against a 'benchmark' habitat state.	Habitat	Parkes et al., 2003
3. UK (biodiversity offset pilot)	A compound calculation method; interchangeable 'units' of biodiversity, calculated based on the 'distinctiveness' and 'condition' of the habitat type. Multipliers account for restoration uncertainty.	Habitat	Defra, 2011
4. Canada (fish habitat)	The area and 'productivity' of fish habitat lost	a) Habitat b) Species	DFO, 2002
5. US (conservation banking)	The area of habitat required to support each family group of a protected species	Species	US FWS, 2006
6. Area only	For comparison - compensation of the area damaged (regardless of condition loss)	Species	n/a
7. Modified Victorian	Same as 2, but with site-appropriate condition indicators	Habitat	Expert opinion

7.2.1 *Baseline data on impacts upon biodiversity*

Habitat

In Chapter 6, I showed that vegetation species richness and percentage vegetation cover approach zero within the area directly occupied by infrastructure on the Ustyurt (i.e. roads, pipelines, extraction sites). However, I also found that vegetation is not significantly affected outside of this footprint, including a lack of significant edge effects from e.g. dust deposition. Based upon these empirical data, a binary function was therefore appropriate, and vegetation was treated as either entirely removed (i.e. condition = 0) or untouched (i.e. condition = 1) by industrial activity in the Ustyurt.

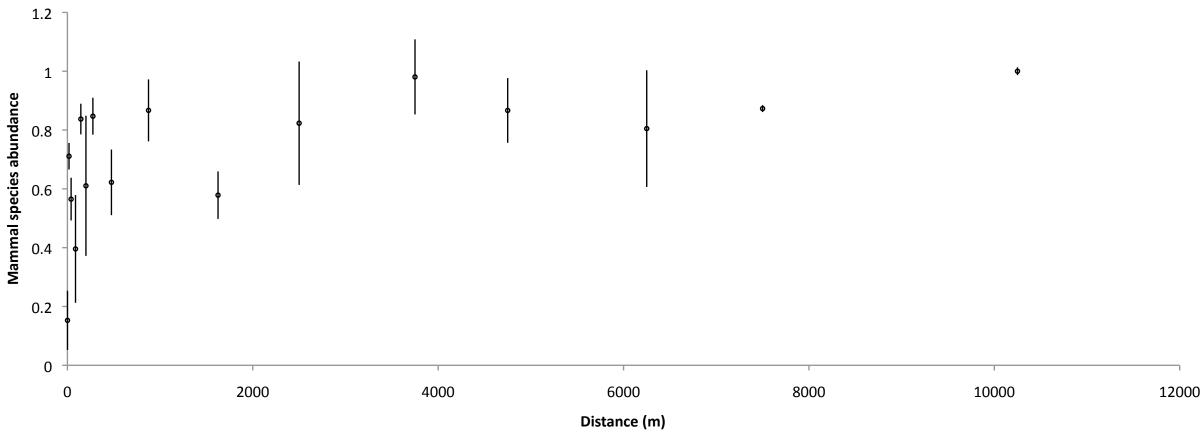
Chapter 6 contains a conservative estimate for the total area of habitat currently cleared by natural gas development on the plateau over the last 40 years, of 220 ± 19 km^2. This figure does not account for any potential fragmentation effects caused by infrastructure across the landscape, but for most roads these are likely to be minimal in relation to the ecology of the region (J. Bull, pers. obs.). I use the total estimate from Chapter 6 to calculate habitat offset requirements across the landscape. It should be noted that in reality, for the Ustyurt plateau, these losses are historical and therefore not likely to now be compensated for under any new offset policy. I do not suggest here that an offset policy should account for historical losses (although this has been proposed elsewhere; Habib et al., 2013), but instead seek to use historical losses in the Ustyurt study to compare and contrast existing international offset policies under a common case study.

Species

I estimated fauna disturbance using a meta-analysis of disturbance caused to mammals by infrastructure (Benitéz-Lopez et al., 2010). This meta-analysis was for a range of species and habitats and is not specific to the fauna of the Ustyurt, but in the absence of peer-reviewed local data, is one of the best available estimates of the effects of infrastructure. I note that in designing an actual offset scheme for the Ustyurt, using such an estimate would be insufficient to capture disturbance to the full assemblage of mammals, let alone other components of biodiversity (e.g. reptiles, birds, invertebrates). However since this is more for theoretical exploration, and the flagship species for conservation in the region is the saiga, the use of a mammalian indicator is logical.

The presence of major infrastructure generally has significant impacts on mammal species abundance up to ~ 5 km from the disturbance (Benitéz-Lopez et al., 2010; Osti et al., 2011; Fig. 7.1). In a similar habitat and for a comparable assemblage of ungulates (the Gobi, Mongolia), a recent report in the grey literature notes avoidance of infrastructure up to at least ~ 1 km away (Huijser et al., 2013). This adds weight to my assumption that using a 5 km disturbance buffer based on Benitéz-Lopez et al. is of an appropriate order of magnitude for disturbance to saigas in the Ustyurt.

Figure 7.1*: Plot of mean normalized mammal species abundance (MSA; taking values between 0 and 1) against distance from infrastructure (in metres), using data presented in a meta-analysis by Benitéz-Lopez et al. (2010)*



Industry

Known O&G infrastructure on the Uzbek side of the Ustyurt Plateau consists of 6 major facilities, 3 major pipelines, multiple large off-road tracks, and a railway (Chapter 6). I assume that villages would have existed without O&G development, despite the presence of industrial infrastructure in their vicinity. Natural gas is the main target for the O&G industry in this region (UNDP, 2010a). The development of this infrastructure has occurred over approximately the last 40 years (A.V. Esipov, pers. comm.). During the two decades for which public data exist, Uzbek natural gas production has increased linearly (EIA, 2012; Appendix A2), so I assume that industrial impacts also increased linearly over the whole 40-year period.

7.2.2 *Offset calculations using different methodologies*

I chose to apply a set of established regulatory biodiversity offset methodologies, taken from a recent review paper, to the Ustyurt plateau for existing O&G infrastructure (McKenney & Kiesecker, 2010; Table 7.11). The Brazilian Forest Code was excluded as not only is its future unclear (Madsen et al., 2011), but it arguably does not fulfill the criteria of a 'biodiversity offset' policy, as outlined in Chapter 2. In addition, for the sake of comparison with more established methodologies, I included the approach to offsetting currently being developed in the UK (Defra, 2011). I also developed and tested a locally adapted version of the Victorian approach of Australian Native Grasslands, which is under consideration for use in the offset policy currently being developed for the Ustyurt by Uzbek authorities ('habitat hectares'; Parkes et al., 2003). I first outline the steps in each method and then describe how I apply them in the Ustyurt case.

US Wetland Banking

Offset requirements are calculated based upon area of habitat lost, or length of feature for a linear feature such as a narrow waterway (US ACE et al., 1995).

Use in this study: Biodiversity offsets were required to replace the same area of vegetation impacted by development (= 220 km^2; Chapter 6), regardless of the magnitude of the impact.

Victorian Native Grassland (Australia)

This approach built around the habitat hectares metric (HH; Parkes et al., 2003), compares vegetation condition with a benchmark (i.e. same vegetation type in an undisturbed condition). The stages in assessing offset requirements are:

(1) Identify Ecological Vegetation Class (EVC), describing natural (undisturbed) state of site condition components. For the purposes of the Uzbek study I ignored this stage, and treated the plateau as one homogenous habitat type. However, in developing a full offset policy for the region, it would be possible to define different vegetative associations (e.g. an *A terrae-albae – A. salsa – Halyoxylon spp* association).

(2) Assess condition of different ecological categories against pristine EVC condition, giving each a score up to the maximum (Table A3.1). The categories in Table A3.1 are intended to be excluded from the analysis when not relevant to a certain habitat type. So, applying this method for the Uzbek case, the sections for 'large trees', 'tree canopy cover' and 'logs' were excluded, as there are very few true trees on the plateau. The category for 'organic litter' was also ignored, as this is not particularly a feature of the region, as was 'lack of weeds', as invasive non-native weeds does not appear to be a threat in this habitat (J. Bull, pers. obs.). Details of scoring for the other categories are shown in Tables A3.2 – A3.6.

(3) The assessment in step 2 provides the 'habitat score'. This is multiplied by the area of the habitat patch to get the score for the area in 'habitat hectares'. Losses and gains can then be compared using the HH score, measured against a benchmark example of that habitat in 'pristine' condition.

Use in this study: to calculate losses, the area of land impacted by industrial activities (220 km^2; Chapter 6) was assumed to have started at benchmark condition (= 1). I completed all calculations in km^2 for simplicity, rather than converting to hectares, making this equivalent to 220 "habitat km^2" that had been influenced by development. The condition score of impacted land was calculated using Tables A3.1 – A3.6, giving a score of: 5 (understory strata) + 1 (recruitment) + 8 (patch size) + 8 (neighborhood) + 4 (distance to core area) = a condition score of 26/60. Normalizing this, to account for the missing categories, gives a score of 43.3/100. Multiplying this by 220 km^2 gives the result that the loss in "habitat km^2" from development is 12,466.6, which is the loss that needs to be offset. The offset requirement, as per my assumptions, can be treated as the area of habitat elsewhere in the plateau currently at a condition level of 0 that needs to be returned to benchmark condition, which is therefore 124.6 km^2.

Adapted Victorian Native Grassland (Australia)

I created a slightly adapted version of the HH methodology with components that were more specific to the Uzbek case study. All categories from the application of existing methodology were retained. Instead of the 'Lack of weeds' section, I used cover of black lichen (*T. desertorum*), which is a sign of undergrazing and prevents vegetative recruitment (G. Gintzburger, pers. comm.) (Table A3.7).

Use in this study: this version of the methodology was applied exactly as for the existing standard version (Parkes et al., 2003), with the exception that the new category for lichen cover was included in the calculations.

UK Biodiversity Offset Pilot

Taken from the guidance published by Defra (2011), this is the offset metric currently employed in the UK pilot scheme. The process is based upon condition and area, incorporating a set of multipliers, and follows the following steps:

(1) Assign habitat type band based on distinctiveness, as classified in the appendix to the Defra guidance – High, Medium or Low.

(2) Score distinctiveness (Table A3.8).

(3) Apply weighting for habitat condition (Table A3.9); note that the approach to weighting habitat condition is currently not defined, but is expected to rely upon guidance in existing agri-environment schemes.

(4) Combine distinctiveness and condition (by multiplication) to give number of biodiversity units per hectare (Table A3.10). The number of units per hectare is then multiplied by the total number of hectares impacted, to give the total number of biodiversity units that need to be offset.

(5) Establish multiplier for category of delivery risk (Table A3.11). Risk is evaluated based upon the technical appendix developed for this purpose (Defra, 2011). None of the habitat types in the UK correspond directly with those in the Ustyurt, although see Gintzburger (1987), where seedling viability increases from 9 to 60 % with restoration of semi-arid desert. I assume that this represents ~ 50% restoration success, and assign a restoration level of Medium difficulty.

(6) Establish multiplier for location of offset, whether it is in the local biodiversity strategy area or not (Table A3.12).

(7) Establish temporal multiplier to account for e.g. delays in restoration. Again, these are defined in the technical appendix (Defra, 2011). None of the habitat types here correspond with those in the Ustyurt, so I assume about 3 – 5 years until maturity with restoration, and > 40 years without (G. Gintzburger, pers. comm.).

(8) Apply the multipliers from steps (5 – 7) to the biodiversity units calculated, and the final total is the number of biodiversity units required in the offset.

Use in this study: I assigned the pre-development Ustyurt habitat a Distinctiveness score of 6 (high) and a Condition score of 2 (moderate). This gives 12 units per hectare under this methodology, or 1,200 units per km^2. Again, I consider the area of habitat affected by development = 220 km^2. Given that the vegetation is effectively cleared over this area, this suggests that there are 1,200 * 220 = 26,400 units of habitat lost through development. The multipliers I apply to this total are 1.5 (restoration risk; Table A3.11), 1 (spatial risk, Table A3.12), and 1.2 (temporal risk; Table A3.13), meaning that 475,200 habitat units are required under offset schemes. Assuming that offset projects would need to take land of zero condition and restore it to Moderate condition (i.e. the benchmark for the habitat type), they would need to realize 1,200 units per km^2, which would necessitate 396 km^2 of offset project.

<u>Canadian Fish Habitat</u>

The objective for Canadian fish habitat offsets is NNL in "productive capacity" of fish habitat (DFO, 2012). Although it is not mandated, the DFO suggest using an established method (Minns et al., 2001) and then applying multipliers to deal with restoration uncertainty and time lags. The theoretical basis for Minns et al. is the application of the equation:

$$\Delta P_{NOW} = [p_{MOD} - p_{NOW}].A_{MOD} - p_{MAX}.A_{LOSS} + [p_{COM} - p_{NOW}].A_{COM} \tag{7.1}$$

where:

ΔP_{NOW} = Net change of natural productivity of fish habitat
A_{LOSS} = Area of habitat lost due to development activity
A_{MOD} = Area modified, directly and indirectly, as a result of the development activity
A_{COM} = Area created or modified elsewhere to compensate for the development activity
p_{MAX} = Maximum potential unit area productivity rate (or productive capacity)
p_{NOW} = Present unit area productivity rate
p_{MOD} = Modified unit area productivity rate in affected areas
p_{COM} = Compensation unit area productivity rate in affected areas.

ΔP_{NOW} is required to be > 0. p_{MAX} is set to 1, and all others are proportions of this.

Use in this study: To achieve NNL, I rearranged equation (7.1), setting $\Delta P_{NOW} = 0$. Consequently, I required that: {area * productivity of lost habitat} + {area * change in productivity of modified habitat} = {area * change in productivity of offset habitat}.

For the purposes of this exploration, the method could be applied to habitat condition for mammal species in the Ustyurt. Alternatively, it could be applied to the quality of the habitat in terms of vegetation. As a result, in this exploration, I use the Canadian methodology as both a habitat-based and a species-based metric. For measures of productivity, I used the proxies of '% vegetation cover' and 'mean species abundance', for vegetation and fauna respectively. The values for all parameters were as listed in Table A3.14.

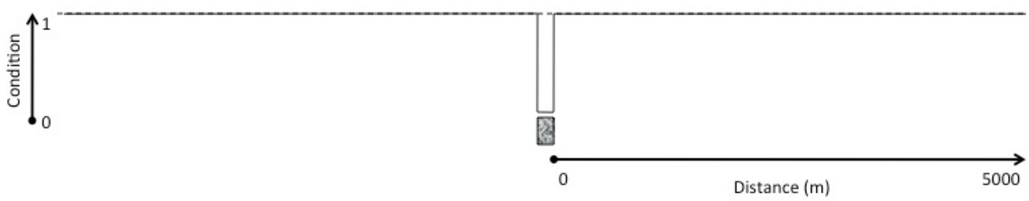
Species-based (US)

There is no standard methodology for calculating credits (i.e. offset requirements) for Conservation Banking. Apparently, “In its simplest form, one credit will equal one acre of habitat or the area supporting one nest site or family group”, and, “…the credit system for a conservation bank should must be expressed and measured in the same manner as the impacts of the development projects that will utilize the bank” (FWS, 2003).

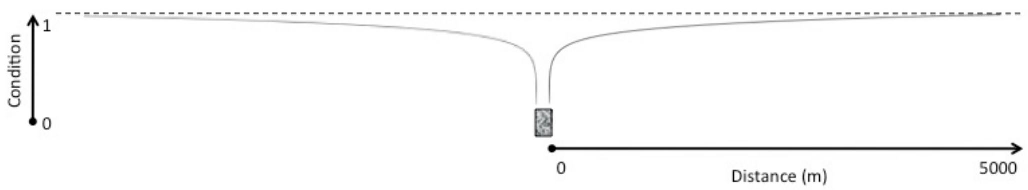
Use in this study: I take this criterion to mean that development impacts result in equivalent area of offset habitat required. Here, my exploration bases the analysis upon mean species abundance (MSA) of mammals. I assume that within the footprint of cleared habitat on the Ustyurt, MSA (and hence condition) is 0, and that it improves outside of this footprint according to the functional form described in the next section (Fig. 7.2). I then split the footprint into its separate ‘linear’ and ‘hub’ components *sensu* Chapter 6, and calculated the additional area that would be disturbed under the MSA measure for each component, using simple geometry. This results in an additional 312 km^2 that had condition reduced from 1 to 0, using the MSA proxy, which when added to the original 220 km^2, gives 532 km^2 of impact. The offset requirement is to fully compensate for this, by restoring 532 km^2 of the landscape from a state at which it has 0 MSA to the benchmark level.

***Figure 7.2**: Functional form of industrial impact upon condition for (a) vegetation, based upon data collected in the field (Chapter 6); (b) mammal species abundance, based upon the outcomes of a meta-analysis (Benitéz-Lopez et al., 2010). Grey block = industrial activity; dashed line = benchmark condition; solid lines = functional form of condition*

(a)



(b)



7.3 Theory – implementation of the metrics

7.3.1 *Functional form of industrial impacts*

I call the change in condition with distance from industrial infrastructure the 'functional form' of impact. Estimating and using the functional forms of impact could allow a better estimation of the residual biodiversity impacts associated with development than simply using buffer zones. Functional forms were generated for both vegetation impacts and mammal disturbance, to allow estimation of losses in these biodiversity components caused by development. In Chapter 6, I found that vegetation was completely cleared (i.e. condition approaching 0) on the site of infrastructure, but not significantly affected outside of this direct footprint (i.e. condition approaching 1), so the functional form for habitat was treated as a step function (Fig. 7.2a).

The data used to derive a functional form for mammal disturbance (Fig. 1) had a good fit (r^2 = 0.63) to the relationship MSA = 0.0693.ln[x] + 0.2936, where MSA is 'mammal species abundance' and x is distance (Fig. 7.2b). The definite integral of this relationship between x = 0 and x = 5000 (metres) gives the condition-area under the curve over that distance (CA_{below}). The value of MSA at a hypothetical control site (condition = 1) multiplied by the same distance (x = 5000 m) gives the benchmark condition area CA_{max} = 5000. The amount of condition-area lost as a result of the presence of infrastructure can then be estimated as: $(CA_{max} – CA_{below}) / CA_{max}$

= 233.7. In terms of area under the curve, this would be equivalent to a step function in which MSA = 0 from the point of disturbance as far as 233.7 m, and MSA = 1 from 233.7 m outwards.

7.3.2 *Biodiversity offset projects*

Offset projects require the creation of additional biodiversity value (BBOP, 2012), and so those hypothetically implemented in my calculations needed to raise the condition of degraded land in the Ustyurt. I note that in the case of offsets more generally, it is possible in deteriorating landscapes to implement offset projects that prevent biodiversity losses that would otherwise have occurred i.e. 'averted loss' offsets (Gordon et al., 2011a; Chapter 4). It appears that the Ustyurt habitat can be said to have deteriorated in recent decades, as discussed at length in Chapter 5. However, to concentrate on the comparative study of different metrics, I simplify the case by treating the habitat as stable.

In practice, habitat-based offsets might involve managed habitat restoration, such as reseeding areas in which vegetation had previously been cleared, or planned grazing of under-grazed areas to reduce lichen growth. For species-based offsets, activities might include reductions in non-O&G related disturbance, e.g. poaching activity. Offsets would not be implemented on the same site as new or contemporaneously operational developments, but elsewhere in the plateau. For instance, species-based offsets might be strategically implemented in the far north of the Uzbek plateau, where there is less extractive activity but relatively high saiga density (Chapter 5).

The offset requirements were calculated on the highly simplified basis that all offset projects take land somewhere in the plateau that has zero condition, but no existing infrastructure, and restores it to pristine levels of vegetation cover or suitability for mammals (i.e. condition = 1). The basis for this from a habitat point of view is that the scrub vegetation could be completely cleared in a patch and consequently treated as of approximately zero condition, but with reseeding could feasibly be made indistinguishable from an untouched area (i.e. condition approaches 1) after a few years. From a species point of view, the suitability of a patch for mammals as characterized by the flagship saigas might be determined by how much hunting takes place in the patch, such that there could be areas where almost all mammals are hunted (i.e. condition approaches 0), but that with a concerted effort to reduce hunting through offset projects, becomes essentially safe for mammals (i.e. condition approaches 1). It was assumed that suitable areas for restoration were not in limiting supply; given that the amount of habitat cleared for current oil and gas infrastructure has been estimated at <1% of the region by area, this is not an unreasonable assumption.

Summing estimated biodiversity losses and potential offset gains, on development and offset sites only, over the assumed 40-year period, gives a net biodiversity outcome against a project-scale baseline considering development and offset sites only, rather than the wider landscape

(Chapter 4). I considered both in-kind offsets (restoration with the same target as the loss) and 'out of kind' offsets (restoration of a different target as the loss). For example, the loss of an area of vegetation impacted by infrastructure could be compensated for by reseeding of the same area of condition zero vegetation elsewhere, where condition zero means bare soils (in kind) or by paying for anti-poaching patrols elsewhere, where condition zero means the flagship species is heavily poached (out of kind).

To calculate the net area of benchmark condition that would have been gained had 'out of kind' offsets been implemented, I took the difference between the area of mammal habitat gained and the area of vegetation lost. A 5-year maturation period was assumed for vegetation restoration activities, but mammal protection measures were assumed to act within a year.

Some methodologies include correction factors for uncertainty. I included two types of uncertainty in my calculations of net biodiversity outcome: the uncertainty in the amount of land impacted by development (from Chapter 6), and the possibility of up to 50% non-implementation of the offset policy on the ground. The latter is an arbitrary but realistic rate (Chapter 2). Uncertainty arises from numerous other sources that I do not consider here, some of which are potentially more important in terms of absolute outcomes (Moilanen et al., 2009; Kujala et al., 2012). An exhaustive framework for quantifying uncertainty in offset projects has yet to be developed (Chapter 2), although tools for managing uncertainty are under development (e.g. Pouzols et al., 2012). The aim of my exploration was to focus on a comparative study and not upon uncertainty, but I consider it to this limited degree for two reasons: firstly, to demonstrate how it can have a significant influence on comparative outcomes, and secondly, to highlight that it is a topic that cannot in general be ignored in modeling conservation interventions.

7.4 Results

7.4.1 *Total offset requirements*

The offset requirements calculated using different methodologies varied quite substantially (Table 7.2; $n = 6$, $\mu = 338$ km^2, $\sigma = 174$ km^2, $CV^* = 1.29$; where μ = mean offset requirement, σ = standard deviation, CV^* = unbiased estimator of Coefficient of Variation to account for small sample size). Incidentally, in completing the same calculation, but incorporating the large estimate for the amount of mammal habitat that is influenced by the development and ignoring the functional form of disturbance (= 9,023 km^2), the variation is much larger ($n = 7$, $\mu = 1{,}578$ km^2, $\sigma = 3286$ km^2, $CV^* = 5.73$). However, it is already widely accepted that area alone should not be used as a measure of biodiversity losses and gains in offset schemes (BBOP, 2012).

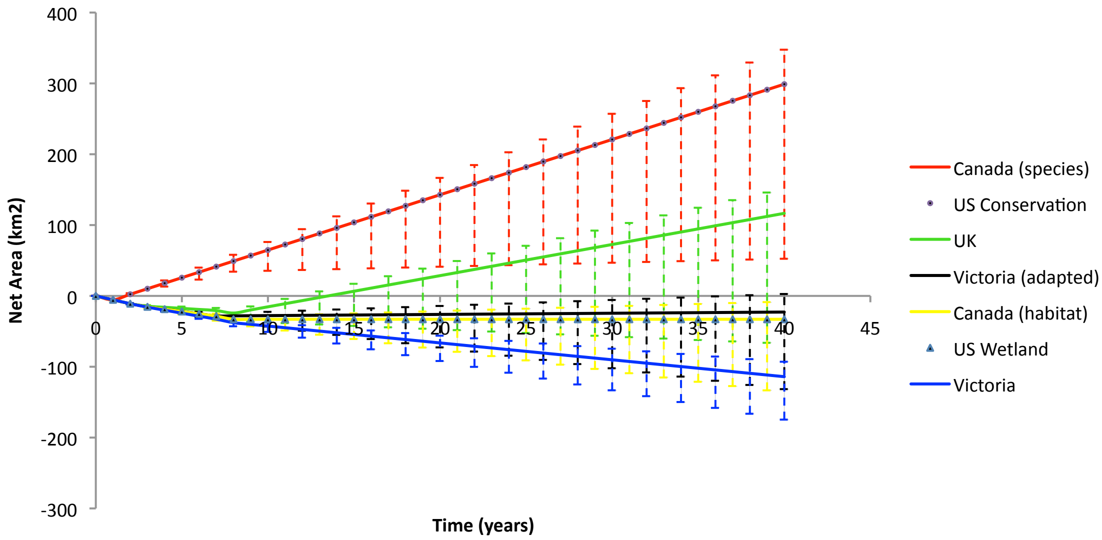
***Table 7.2**: Comparison of offset requirements by area to which additional conservation actions need to be applied, under a static appraisal of 40 years of O&G development, using different offset methodologies. Uncertainty represents the potential range of spatial extents of oil and gas infrastructure across the Uzbek Ustyurt estimated in Chapter 6. 'Net Area' is the difference between the required gains ('Area' column) and the area impacted after 40 years of O&G development (220 ± 19 km^2; Chapter 6). The "saiga habitat" row shows the area of the proposed saiga reserve that could be used for restoration of habitat condition under species-based offsetting, to give context.*

Offset policy	**Target**	**Area [km^2]**	**Uncertainty [km^2]**	**Net Area [km^2]**
1. Area only (US)	Habitat	220	± 19	- 33
2. Area and condition (Victoria)	Habitat	125	± 11	- 114
3. Area and condition (UK)	Habitat	396	± 34	+ 116
4a. Area and functionality (Canada)	Habitat	220	± 19	- 33
4b. Area and functionality (Canada)	Species	532	± 46	+ 299
5. Area and condition (US)	Species	532	± 46	+ 299
6. Area only	Species	9,023	± 779	n/a
7. Area and condition (Victoria adapted)	Habitat	227	± 20	- 22.8
Proposed Saigachy reserve	Saiga habitat	7,352	n/a	n/a

7.4.2 *Offsetting through time*

The various habitat-based methodologies result in a range of positive and negative net outcomes, with none resulting in exactly NNL (Fig. 7.3). Most result in a net loss, although this is partly due to the assumption of a time lag between development impacts and offsets maturing. The UK calculation method initially results in a net loss, but after 12–13 years of development attains NNL, and then goes on to result in a net gain; but only if full compliance is achieved. The Victorian calculation method delivers insufficient compensation and so results in increasing net loss. The adapted Victorian calculation method came closest to NNL, although it had still not been achieved after 40 years.

Figure 7.3: *Plot of net area of land at benchmark condition (in km^2) against time (in years) as a result of hypothetical natural gas offsets in the Ustyurt, using different methodologies (Table 7.1). The Canada (species; 4b in Table 7.1) and US Conservation (5 in Table 7.1) methods are exactly aligned, and represent 'out of kind' offsetting in this case. Upper and lower bounds reflect uncertainty in both estimation of impacts (cf Table 7.2) and, for the lower bound, the possibility of up to 50% non-compliance.*



The habitat calculation methods generally differ from the species based methods, as expected. The Canadian and US Conservation Banking calculation methods both suggest the same offset requirements for habitat and species, but the out of kind offsets resulted in a net gain in the area of benchmark condition land. This net gain arises because gains in species habitat are more diffuse than the highly localized losses in vegetation habitat, so a wider area is modified through offset activities.

7.5 Discussion

7.5.1 *Comparing different offset metrics*

The different methodologies resulted in a range of required ecological gains. Under the assumptions made here, none resulted in NNL, on the basis of the metrics that I have chosen and against a fixed baseline. The divergent outcomes are likely to be due to: (a) differences in interpretation of ecological equivalence; (b) the calculation method being overly specific to national habitats and therefore not adequately capturing Uzbek biodiversity; (c) multipliers being explicitly built into calculation methods; and (d) the assumed time lag in habitat restoration gains.

The specificity of the Victorian calculation method to Australian native grassland explains the low restoration estimate for the non-adapted version of the method. In particular, the Uzbek plateau scores highly under this calculation method for a lack of invasive weeds, which is an important problem for Australian grassland but much less so for Uzbek vegetation. Whilst this is perhaps not surprising, given that the method was designed specifically for Australia, it is nevertheless a pertinent point because Uzbek policymakers are currently considering the applicability of the habitat hectares metric in the Ustyurt (J. Bull, pers. obs.). The UK calculation method required a larger area to be offset due to the enforced use of multipliers, which are intended to account for uncertainty in implementation (Moilanen et al., 2009). Recent work has found that high multipliers may be necessary to achieve NNL in practice (Pickett et al., 2013). Conversely, the US and Canadian calculation methods are more open to interpretation, to the extent that the method could be applied to both habitat and species.

As my results have emphasized, the Victorian metric is extremely specific to the habitat it is designed for, and should not be used elsewhere without adaptation. The modified version, suitable for the Ustyurt habitat, came the closest to achieving NNL using the metrics and under the assumptions that I have specified here. Rather than rushing to implement this calculation method in practice, though, a further exploration would be necessary of the uncertainties related to the raw data used, impact quantification, distribution and mobility of the focal species, climatic change and uncertainty related to the implementation and governance of the offset.

Although it is not something I have modeled here, the difference in the degree of flexibility between offset methodologies suggests that some may have more uncertain outcomes than others, or at least be characterized by additional potential sources of uncertainty. This is important, given that even the limited treatment of uncertainty I include here results in a large overlap between otherwise divergent outcomes (Fig. 7.3). An interesting further study would be to repeat my analysis using a tool such as RobOff, which allows users to optimize conservation actions given a more detailed consideration of uncertainty (Pouzols et al., 2012).

The divergence in outcomes is informative regarding any debate on international trade in biodiversity credits. Despite the common NNL objective, one difficulty that would be encountered would be demonstrating equivalence between credits generated in different places, under different offset systems. Whilst international credit trades have yet to be officially proposed, the concept is a common topic of conversation and debate amongst offset researchers and practitioners (J. Bull, pers. obs.). I do not explore the topic further here, rather noting that it is a topic for which my results have some relevance.

7.5.2 *Out of kind offsetting*

It might be argued that restoration of vegetation in the Ustyurt is not the most urgent conservation priority in the region. Not only does the area impacted by industry constitute much less than 1% of the Uzbek part of the plateau (Chapter 6), the vegetation in the region is undergoing wider decline in any case as a result of the drying of the Aral Sea (Micklin, 2007). As such, small-scale restoration efforts would do little for the habitat as a whole. An alternative would be to consider out of kind offsetting, as captured here by the examination of species-based offsets (US and Canada).

We have not here developed a scale for equivalence between losses in vegetation and gains in undisturbed mammal habitat (cf Quétier & Lavorel, 2012). Such a scale might enable the out of kind trading of these different components of regional biodiversity; and it is not hard to imagine that one could be conceived. Trading vegetation losses for gains in threatened species conservation in the Ustyurt could potentially result in a net gain from a conservation point of view, as the latter is a far more urgent conservation priority (particularly saigas; Milner-Gulland, 2010). An exploration of relative costs would be required to ascertain which approach was likely to be more cost-effective. Recent studies in different ecosystems have shown that out of kind offsets can result in more efficient use of conservation funding (Habib et al., 2013), so it is not unfeasible that a similar outcome would be achieved for a case such as the Ustyurt.

Out of kind trades require acceptance that funding paid in direct compensation for biodiversity losses could be utilized to address different conservation priorities. This is possible under some offset schemes (McKenney & Kiesecker, 2010) and encouraged under others (Brownlie & Botha, 2009) but risks blurring the line between what might be considered 'strict' biodiversity offsets and straightforward fines or taxes for environmental damage. However, absolute gains might be much larger if the offsets focused on species or assemblages of particular conservation concern rather than generic vegetation or habitat loss. One key driver of species decline in the region, especially the flagship saiga antelope which would be the focus of any offset scheme (UNDP, 2010a; Chapter 3), is poaching by those not involved in the O&G industry (Kühl et al., 2009). Without any offset schemes, this decline is likely to continue, but if offsets are used to prevent poaching, populations could perhaps recover. This would arguably represent a much better outcome for conservation than adhering to a strict 'like for like' NNL framework or not doing anything at all – and is something I explore in the next Chapter.

Choosing the most appropriate calculation method for an offset scheme from a divergent set is clearly about much more than simply selecting characteristic or representative components of the ecosystem in question to measure; it also requires a clear decision as to the fundamental objective of the offset policy.

Chapter 8

Combining flexible and non-flexible biodiversity offsets to achieve improved conservation outcomes

"You delight in laying down laws,
Yet you delight more in breaking them…
But what of those to whom life is not an ocean, and man-made laws are not sand-towers,
But to whom life is a rock, and the law a chisel with which they would carve it in their own likeness…
What of the ox who loves his yoke and deems the elk and deer of the forest stray and vagrant things?"

Kahlil Gibran (from "The Prophet", 1923)

"I don't have much money, so I make sure it's the expensive kind."

John Bull (from "Hearse Car Blues", 2011)

8.1 Introduction

A commonly held policy principle of biodiversity offsetting is that the biodiversity gains associated with offset projects should be in kind wherever possible, i.e. of the same type as the losses for which the projects compensate (Quétier & Lavorel, 2012; Pilgrim et al., 2013). However, there have been recent calls to explore in more detail the opportunities provided by so-called flexible offsets, i.e. those in which the exchange of 'out of kind' losses and gains is permitted (Overton et al., 2012). In the case of flexible offsets, losses in one component of regional biodiversity (e.g. vegetation) might be exchanged for gains in another (e.g. protected areas for endangered fauna), using some kind of equivalency scale (Quétier & Lavorel, 2012). Or, alternatively, losses might be compensated for with gains that are a long distance away from the associated development site, e.g. such as offsets proposed to compensate for seabird bycatch in marine fisheries by removing invasive predators from seabird breeding colonies elsewhere (Wilcox & Donlan, 2007). It has been proposed that flexible offsets can allow for more cost-effective ways of spending available conservation funding (Habib et al., 2013).

In this Chapter, I extend the existing research into flexible offsets by applying them to moving conservation targets. Whilst flexible offsets have been demonstrated to be useful in static analyses (Habib et al., 2013), it is not clear if this is also the case when flexible offsets are implemented in the context of ongoing environmental change. Further, there has been no exploration in the literature of the outcome of combining flexible and non-flexible offsets. I explore these themes for my primary case study: offsets for extractive activities in the Ustyurt plateau, Uzbekistan. Deciding to consider flexible offsets in the first place is a value judgment, and is controversial. However, the use of flexible offsetting in the Ustyurt case has some intuitive logic if the objective is to achieve the best possible outcome for conservation: vegetation loss due to industrial activity is < 1% of the habitat by area (Chapter 6), and so losses and gains would arguably be an immaterial issue for this component of biodiversity. Conversely, funding for saiga conservation could result in important gains for the flagship conservation target in the region, and a target which is perhaps proving more susceptible to current drivers of ecosystem change (Chapter 5). Further, offsets based upon saiga protection could be designed around the more mobile biodiversity conservation interventions suggested earlier for the Ustyurt in this thesis (Chapter 3). These include: implementing protected areas for saiga that are coordinated across their range, implementing mobile or temporary protected areas, encouraging alternative livelihoods, enforcing saiga impact mitigation measures at development sites, and creating protected areas that incorporate measures to be resilient (e.g. core areas, connectivity).

I apply two open-access software programs developed for the purposes of conservation planning under uncertainty: RobOff (Pouzols & Moilanen, 2013) and Zonation (Moilanen et al., 2005). RobOff is a non-spatial decision support tool that is used to find the most desirable mix of different conservation actions that could be applied to a landscape over time, given uncertainty in the responses of biodiversity features to each action, and the costs of each action within a fixed budget. Whilst the program was developed recently, it has been trialed (Pouzols & Moilanen, 2013) and used in practice to explore approaches to conserving Australia's wildlife in the face of climate change (Maginni et al., 2013), though not yet for biodiversity offsetting. Zonation is better established, having been in development since 2003 (Moilanen et al., 2012). It is a framework for large-scale spatial conservation planning, used to identify areas within a wider region that are priorities for multiple different biodiversity features, given consideration of the connectivity between areas. RobOff and Zonation are both applied to the available data for the Ustyurt, to answer a set of related questions about the potential for flexible offsets in the region.

The key questions are, firstly, whether flexible offsets (as categorised in the section 8.2) offer a good alternative to non-flexible offsets for conservation in the Ustyurt, when considering the mobile nature of conservation targets. The second is where offsets should be located within the Ustyurt, given the specified conservation targets. Thirdly, I evaluate the performance of a combination of flexible and non-flexible offsets over a range of timescales up to 100 years. The results obtained relate specifically to the case study in question, but allow a more general discussion of the potential merits of flexible offset mechanisms in dynamic systems, and of the forms that flexible offsets can take.

8.2 Methods

The study region under analysis is the section of the Ustyurt plateau contained within the national borders of Uzbekistan and the edge of the plateau to the east, as defined earlier in this thesis (Chapter 5). The defined study area is the same region for which an offset scheme, designed to compensate for losses associated with oil and gas activities, is being developed. It has already been suggested that compensation, arising from offsets for habitat impacts around industrial infrastructure in the case study region, may be more usefully directed towards the conservation of charismatic fauna in remote regions of the plateau rather than in non-flexible offsets close to development sites (UNDP, 2010a). Further, the Ustyurt is a system known to have undergone rapid recent change, as a result of both the drying of the Aral Sea and socioeconomic development. The plateau is also characterized by a migratory flagship conservation target – the saiga antelope (Chapter 3).

8.2.1 *Biodiversity losses and gains*

The key data sets available for the case study region that are relevant to the current study are as follows: a map of localised trends in normalized difference vegetation index (NDVI) for the Ustyurt plateau (2002 – 2012); spatially referenced participatory saiga monitoring data (2006 – 2012); and the presence of natural gas infrastructure constructed over a period of approximately 40 years (Chapter 6). The NDVI trend data are an imperfect proxy for change in vegetation cover, and the other two data sets are subject to large uncertainty, however, they represent the best available existing information relevant to conservation planning in the study region.

For this case study, I consider flexible offsets as those in which ecological compensation is not constrained to being of the same type as losses associated with extractive development (i.e. flexibility in kind), or in which compensation is not constrained to being near to that development (i.e. flexibility in space). Biodiversity losses from development are considered to be decreased vegetation cover and increased disturbance to habitat for mammals in general. Biodiversity gains to be achieved through offset projects can either take the form of restored vegetation or areas of reduced disturbance to mammals (non-flexible offsets), or can alternatively be achieved through reduced disturbance to saigas specifically, via e.g. controlling poaching (flexible offsets).

Biodiversity losses are calculated based upon the spatial footprint of oil and gas infrastructure in the plateau. This means that losses can be considered to be 220 ± 19 km^2 of vegetation and 532 ± 46 km^2 of habitat for mammals, over a 40 year period (Chapter 6). Biodiversity gains are required for the same absolute area as losses but are multiplied by an arbitrary factor of 10 (Moilanen et al., 2009). A multiplier of this order of magnitude was shown to be necessary for achieving no net loss in practice, albeit for a very different case study (Pickett et al., 2013).

Spatial trends in spring NDVI over the last decade were used to prioritize locations for vegetation restoration (non-flexible) offsets. Priority areas were considered to be those where NDVI has changed the most, the rationale being that NDVI in the Ustyurt is known to be relatively stable (Chapter 5), and hence large changes in NDVI could potentially relate to some undesirable modification of green vegetation cover on the ground. This is recognized to be a large assumption, but in the absence of a better understanding of the link between NDVI, vegetation and conservation priority in the Ustyurt, it at least provides a basis for exploring the issues around flexibility in offsets. The locations at which saigas have been observed during the period 2006 - 2012 were treated as priority areas for offsets that improve the suitability of habitat by reducing poaching rates (i.e. flexible offsets). Whilst the saiga monitoring data has certain limitations, such as being presence-only data with no estimation of monitoring effort, it is considered to provide a rough overall estimate of the extent to which saigas use different areas of the Uzbek Ustyurt. Saigas are persecuted throughout their range on the plateau, so it can be assumed that those areas in which the presence of saigas is more likely offer more opportunities for anti-poaching initiatives.

8.2.2 *Should flexible offsets be used: RobOff*

I first specify a set of 'environments' (i.e. types of habitat within the landscape), 'features' (i.e. biodiversity features of interest) and management 'actions' to be applied to these features by RobOff. Subsequently, the likely 'responses' of each feature, within each environment, to each action, are specified – along with the uncertainty envelope for the response.

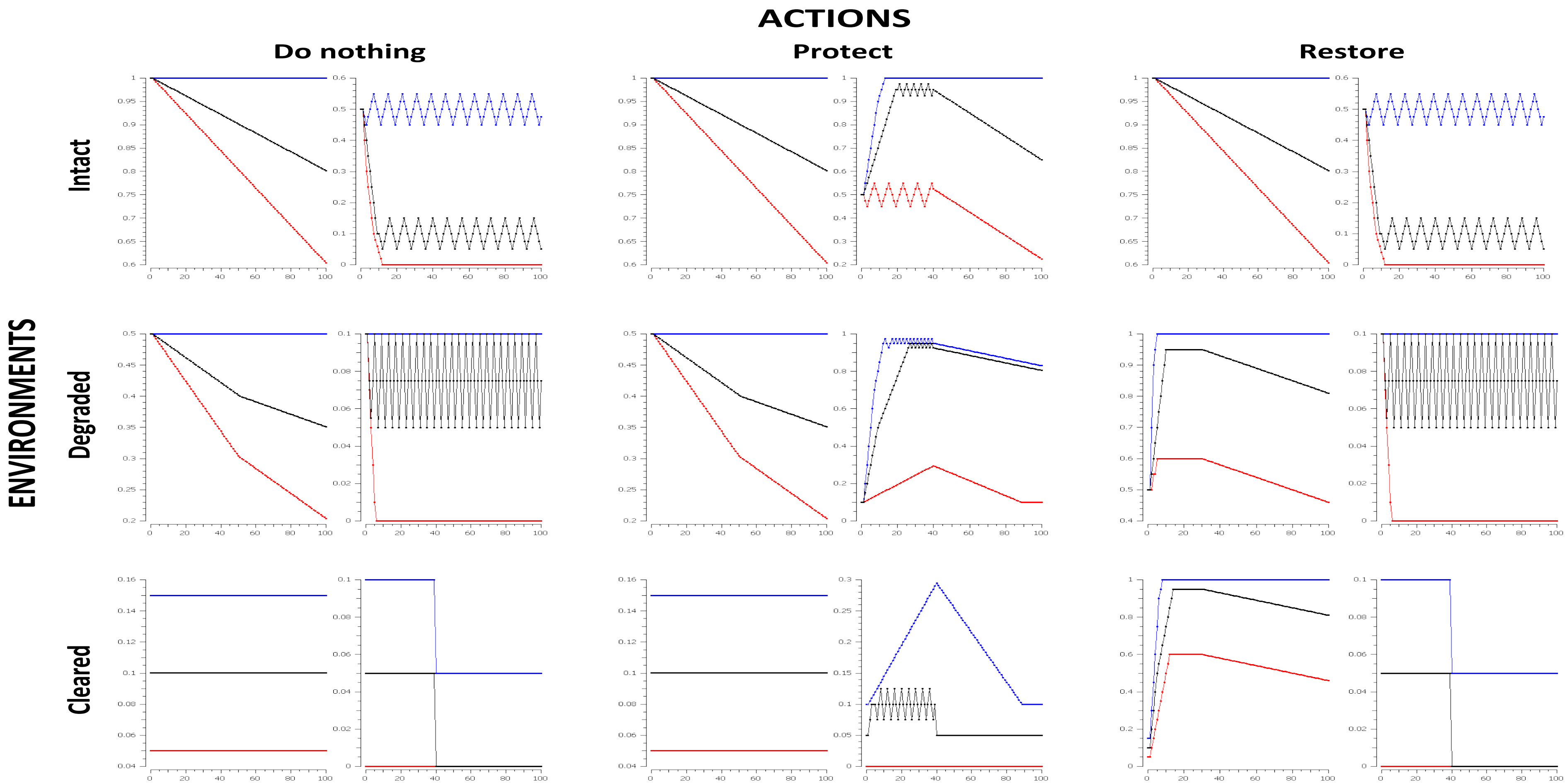
I make the simplifying assumption that the Ustyurt landscape is essentially one continuous broad habitat type (cf Chapter 7), and divide the landscape into three environments: regions that are 'intact' (~ 49.5% of the plateau by area), 'degraded' (~ 49.5%) or 'cleared' (< 1%). The area represented by cleared environments is taken from Chapter 6, whilst in the absence of quantitative data, the remaining area is split equally into intact and degraded environments – which approximately corresponds to observations in the field. The main features of conservation interest in the landscape in relation to offset schemes are considered to be *f1* (green vegetation) and *f2* (saiga antelope habitat) (cf Chapter 5). These two features are each assigned a score of between 0.00 and 1.00 per unit area (km^2), within each of the three environments. For *f1* (i.e. vegetation), a score of 1.00 is considered to be the proportion of green vegetation cover that would be observed in a completely undisturbed area of the plateau. For *f2* (i.e. saiga habitat) the score is considered the likelihood that saiga antelope will not be poached, which is treated as ~ 1.00 in a completely undisturbed area of the plateau. I consider three possible actions: 'do nothing' from a conservation standpoint (i.e. no vegetation restoration or saiga protection activities, business as usual), protection of fauna (flexible offsets), and restoration of vegetation (non-flexible offsets).

<u>Responses of biodiversity to management</u>

The most likely actual responses of the scores of features *f1* and *f2* to these three types of action are currently unknown. Here I estimate responses based upon a counterfactual that has been developed for the Ustyurt incorporating various trends (Chapter 5), and upon personal experience of the system (Chapter 6). The estimated trends are as follows: in intact environments where poaching is known to occur such as the far north of the plateau, vegetation has a score of 1.00 and saiga habitat a score of 0.50. Under 'do nothing', the vegetation score slowly declines due to the drying of the Aral Sea and climate change, whilst the saiga habitat score declines due to ongoing poaching. Poaching does not eliminate saigas entirely however (i.e. score of *f2* > 0.00), and the score fluctuates due to stochastic variation in poaching effort. In reality, due to the assumed uncertainty envelope (see below), this fluctuation is unlikely to materially affect analytical outcomes – furthermore, it represents large assumptions about system dynamics. However, it is retained in the model in order to highlight the point that saiga poaching would be expected to be highly variable. In degraded environments such as less remote areas, the score for both vegetation and saiga habitat is initially lower (0.50 and 0.10 respectively) but is subject to similar trends under 'do nothing'. There is an insufficient time period for ether to drop to zero (Fig. 8.1). This same reasoning applies to cleared environments such as those near infrastructure, again with lower initial scores than degraded environments (0.10 and 0.05 for vegetation and saiga habitat respectively), except that poaching eventually reduces the value of saiga habitat to 0.00 in these areas after ~ 40 years. The action of 'protection' has no effect on vegetation, and that of 'restoration' has no effect on saiga habitat. 'Protection' initially causes saiga habitat to increase in score by reducing poaching effort, but the score eventually starts to slowly decline again in the long term due to the likelihood of a renewed poaching effort in the future; which is not unlikely, given historical trends (Bekenov et al., 1998). ‘Restoration’ causes a relatively rapid increase in the score of vegetation, but once this action has been implemented, it is assumed that it is not then maintained and score begins to decline as per the ‘do nothing’ action.

This set of projected score responses is clearly based upon large assumptions, but my goal here is not to create a detailed predictive model for the Ustyurt. Rather, it is to develop a conservation strategy given a high degree of uncertainty around the behaviour of the ecosystem. Uncertainty envelopes are estimated and included in the response of each feature above, and they are purposefully large due to the lack of data about feature responses in this case study. RobOff allows all of these feature responses to be represented graphically (Fig. 8.1).

***Figure 8.1**: Experimental setup for use of RobOff software in the Uzbek Ustyurt, showing predicted responses of the score of biodiversity features f1 and f2 to different management actions in different environments (y axes) against time in years (x axes). Y values are between 0.0 and 1.0, but note that not all y axes in the figure are to the limits 0 – 1. Time period is 100 years. In each category, vegetation (f1) is the left hand curve, saiga habitat (f2) is the right hand curve. Black line = predicted trajectory, which falls within blue (upper) and red (lower) uncertainty bounds.*



Budget and costs

The costs for the implementation of these three types of action, and the maximum available budget, were included to allow RobOff to process the best possible mixture of conservation actions. The cost of saiga protection per unit area was estimated based upon a draft budget prepared for the creation of a new Saigachy reserve in the Ustyurt (Esipov et al., 2009; Chapter 3), and set at \$2.95 per km^2 per year. No defined estimate is available for the cost per unit area of vegetation restoration, although it was discussed with experts in Uzbekistan (G. Gintzburger, T. Rajabov, pers. comm.) and it was assumed to be much less per unit area than that of saiga protection, so I assumed a value of \$1.00 per km^2 per year. Given that there exists a proposal to use biodiversity offsetting to fund the Saigachy reserve (UNDP, 2010a), the total cost of establishing this reserve was assumed to represent the maximum possible budget available for offsetting: \$800,000 over 40 years.

Definition of a sustainability ratio

RobOff was used to explore the possible combinations of scores obtained for each feature, as a result of implementing actions in the various environments, within the limit set by the total available budget. Each possible combination of scores is aggregated by feature, and then transformed using a benefit function into a 'conservation value' for each feature (I assume the default benefit function: a convex increase with diminishing returns). A 'global' conservation value is then calculated by aggregating conservation value for each feature, taking account of uncertainties in responses to management. The actual algorithms applied by RobOff are detailed in the manual (Pouzols & Moilanen, 2012). Subsequently, RobOff calculates a 'sustainability ratio', defined as the ratio between the global conservation value obtained when a combination of conservation actions is implemented, and the conservation value that results from no action at all. Given this definition, a sustainability ratio of 1 can essentially be considered as the achievement of 'no net loss' of biodiversity. RobOff is intended to optimize the use of an available budget for conservation in a given region; by this, it is meant that RobOff finds the best possible sustainability ratio that can be achieved respecting budgetary limits within that region. The sustainability ratio is the main indicator I use to report the results, alongside the proportion of the budget spent on different actions.

The sustainability ratio can be calculated either including or excluding uncertainty in the responses of features to actions. A 'nominal' sustainability ratio, as defined within the RobOff package, ignores uncertainty in feature responses and is calculated using only the predicted responses. Conversely, a 'robust' sustainability ratio is one in which the uncertainty envelope, defined by the user for each feature response (Fig. 8.1), is considered. In this case, the sustainability ratio is calculated using the minimum global conservation value that would be obtained within the range of uncertainty. A third alternative is the 'opportunity' ratio, which is calculated using the maximum global conservation value that would be obtained within the range of uncertainty, i.e. if feature responses to actions resulted in higher conservation value than predicted (Pouzols & Moilanen, 2012).

I consider both a 'strong' and 'weak' sustainability ratio, as calculated during analyses. The weak sustainability ratio is that discussed above, which is based upon the global conservation value across all features. The strong version of the sustainability ratio, however, is calculated for the single feature that performs worst in the landscape, i.e. the minimum sustainability ratio by feature. It effectively highlights whether some features are performing much worse than others, in a landscape where 'conservation value' is aggregated across multiple features, and thus provides a means for considering potential weaknesses in flexible offsetting (Pouzols & Moilanen, 2012).

The optimization undertaken by RobOff was considered over a variety of spatial and temporal scales. This was so as to build upon the discussion around using different spatial and temporal frames of reference (Chapter 4). Spatially, the optimization was completed at both a project scale (i.e. development and offset sites included only, up to a maximum of ~ 5,000 km^2, which is approximately the total amount developed and offset when a multiplier of 10 is applied), and at a landscape scale (i.e. the entire Uzbek plateau, ~ 100,000 km^2). Temporally, the optimization was completed using exactly the same set of responses but over different time periods of 5, 10, 25, 50, and 100 years. A discount rate can be applied to sustainability ratios within RobOff to account for time preferences, but for simplicity here I set it to zero and include it as a parameter within a sensitivity analysis. There is still much debate on the correct discount rate to set in such cases, although it would generally be > 0 (Overton et al., 2012).

Sensitivity analysis

Due to the fact that large assumptions were made in estimating key parameter inputs for the RobOff analyses, a simple sensitivity analysis was completed. A list of key input parameters was defined: total available budget (`budget`), cost per unit area saiga protection (`saiga`), cost per unit area vegetation restoration (`vegetation`), area impacted by industry (`impact`), and discount rate (`discount`). The analyses were repeated 50 times using a range of values for each of these parameters, and the resulting weak nominal sustainability ratio, evaluated over a 50-year period, was noted. The database of sustainability ratios and parameter values thus obtained was assessed using a generalized linear model in the statistical package R (R Development Core Team, 2012), and the formula:

`sustainability ratio ~ budget + saiga + vegetation + impact + discount`.

The p-values obtained from a summary of the generalized linear model were considered for significance.

8.2.3 *Where should offsets go: Zonation*

In order to consider how offsets could be spatially distributed in the Ustyurt, I use an open access conservation planning software package, Zonation (v3.1) (Moilanen et al., 2005). Zonation can be used to strategically prioritize areas for protection or restoration based upon the existence of, and connection between, biodiversity features of interest. As such, I utilize it to identify a robust spatial distribution for offset sites in the Uzbek Ustyurt, under different flexibility constraints. Zonation prioritizes regions in a landscape based upon: multiple ecological components of conservation value; the goal of maintaining connectivity; and objectives concerning species' long term persistence. Zonation was chosen for this analysis due to its ability to process raster images containing large numbers of conservation 'sites' (i.e. pixels), and the option to negatively weight components of the landscape in order to explore multiple competing land uses (Moilanen et al., 2012), in this case offset sites and natural gas extraction respectively. Again, the available data sets are assumed to suggest locations which saiga use most frequently, those where NDVI has changed the most dramatically, and those in which oil and gas infrastructure currently exists (Chapters 5, 6). Here, I process these data in Zonation to answer the question of where offsets should go if they involve saiga habitat protection (i.e. flexible offsets), if they involve vegetation restoration (non-flexible offsets), and if they are required to be close to development sites (i.e. non-flexible offsets) respectively.

During a given analysis, each of the two features (i.e. saiga locations, NDVI) was given a weighting to signify its importance to conservation, which varied depending upon the analysis. Due to the simplicity of the analysis and the limitations upon available data, a straightforward weighting was applied: a value of 1 for whichever feature (saiga or vegetation) was being included in offset projects, and a value of 0 for the other. Oil and gas locations were weighted -1 if offsets were not required to be near to development (i.e. flexible offsets, meaning that offsets would be implemented away from infrastructure if possible), or weighted +1 if offsets were required to be near development (i.e. non-flexible offsets). See below for the different scenarios for which analyses were completed. Note that, in reality, offsets would be unlikely to actually fall on top of existing infrastructure sites and would be adjacent to them – but as I am only interested in the approximate location of the offsets at a landscape scale, this was ignored.

Zonation then calculates the total importance of each pixel within the landscape by combining the occurrence of each weighted feature within that pixel. In this context, by 'importance' I mean 'greatest opportunity to be an offset site' – assuming that those pixels that have experienced the greatest change in NDVI, or were visited by the most saigas, are those that present the best offset opportunities. The software does so by calculating the order in which pixels could be removed from the landscape, so as to minimize the loss of weighted biodiversity features with each pixel removed. Again, under the interpretation here, this would mean that the last pixel removed by Zonation would be the one with the greatest opportunity for a biodiversity offset (either vegetation restoration or saiga protection).

I analysed biodiversity offsetting in the Ustyurt for four different offset scenarios: (I) vegetation restoration offsets compensate in-kind for development, under spatial constraints; (II) vegetation restoration offsets compensate in-kind for development, with no spatial constraints; (III) saiga habitat suitability offsets compensate out-of-kind for development, under spatial constraints; and, (IV) saiga habitat suitability offsets compensate out-of-kind for development, with no spatial constraints. These scenarios were, for completeness, combined with another two: (V) the case in which no development was permitted; and, (VI) the case in which development only occurred (Table 8.1). Note that scenario (III) would be unlikely in practice, as saigas tend to avoid infrastructure (Singh et al., 2010a) and thus saiga protection offsets would potentially not be implemented there – but again, it is included for completeness and because my main focus is the difference in spatial distribution between flexible and non-flexible offsets.

Table 8.1*: List of biodiversity offset scenarios simulated, using the previous 40 years of natural gas activity in the Uzbek Ustyurt*

Reference	Scenario	Flexible in type	Flexible in space
I	*vegetation restoration offsets compensate for development, under spatial constraints*	*No*	*No*
II	*vegetation restoration offsets compensate for development, with no spatial constraints*	*No*	*Yes*
III	*saiga habitat suitability offsets compensate for development, under spatial constraints*	*Yes*	*No*
IV	*saiga habitat suitability offsets compensate for development, with no spatial constraints*	*Yes*	*Yes*
V	*'no development' was permitted*	*n/a*	*n/a*
VI	*'development only' was permitted*	*n/a*	*n/a*

8.3 Results

8.3.1 *Implementing a mixture of flexible and non-flexible offsets through time*

The analyses in RobOff suggested that the weak sustainability ratio for the landscape was maximized when a combination of flexible and non-flexible offsets was implemented. When evaluated anywhere over a 0 – 50 year time period, the combination would involve implementing saiga habitat protection (flexible offsets) in intact and degraded environments, and vegetation restoration (non-flexible offsets) in cleared environments. However, if evaluated over a longer time period of 100 years, the assumed responses in biodiversity features (Fig. 8.1) would mean that carrying out vegetation restoration in degraded environments instead resulted in the higher sustainability ratio. The key finding of interest here is not the exact distribution of funds between flexible and non-flexible offsets, as the responses are highly uncertain, but rather the fact that a mixture of offsets can be optimal and that the optimum mixture of actions can change when evaluated over different timescales (Table 8.2).

In all cases, the robust sustainability ratio was < 1, which suggests that no net loss of biodiversity against a ‘no development’ counterfactual was not guaranteed. Further, the robust sustainability ratio decreased if offsetting was evaluated over longer time periods, perhaps in part because over longer timescales the specified offsetting activities had a smaller contribution to make to the biodiversity of the plateau, when the negative underlying trends assumed in the scenario are taken into account. As expected, the sustainability ratio decreased at a greater rate over longer evaluation timescales in a strong sustainability setup than in a weak sustainability setup (Table 8.2). The strong sustainability ratio considers the worst performing biodiversity feature, and the fact that conservation actions only affected one of the two features meant that implementing one action would always result in poor performance of the feature to which it did not apply.

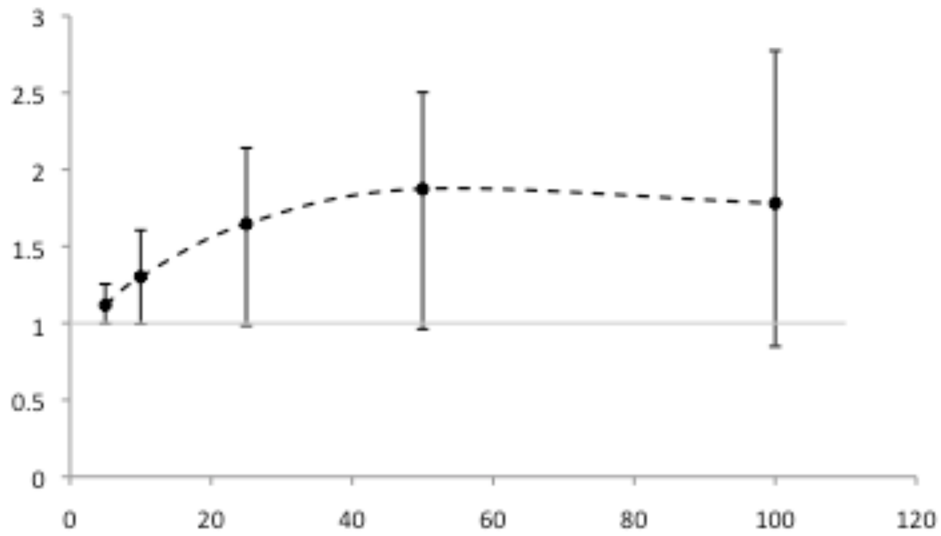
The analyses were repeated at both a project and a landscape scale, *sensu* Chapter 4. However, there was no significant difference between the sustainability ratios achieved across these different spatial scales. This may have been due to the project scale being large enough that it was not saturated with conservation actions.

Table 8.2: *Sustainability ratios (SR) and budget spent ($) on offsetting in the Ustyurt, at a landscape spatial scale, for different time periods. Results given for both a weak sustainability and a strong sustainability experimental setup. Budget rows show spend ($) on saiga habitat protection (P) and vegetation restoration (R) offset actions, for (i) intact, (ii) degraded and (iii) cleared environments.*

LANDSCAPE SCALE		**Years over which evaluated**				
		5	**10**	**25**	**50**	**100**
Weak	SR					
	Robust	0.99687	0.9938	0.98376	0.9616	0.8481
	Nominal	1.1171	1.3018	1.6471	1.8742	1.7803
	Opportunity	1.2546	1.6051	2.1409	2.5026	2.7742
	$ (P:R)					
	(i)	145288:0				145288:0
	(ii)	145288:0				0:49250
	(iii)	0:300				0:300
Strong	SR					
	Robust	0.99044	0.97817	0.9394	0.8732	0.7141
	Nominal	1.0007	1.0014	1.0026	1.0032	1.3572
	Opportunity	1.0107	1.0014	1.0713	1.1607	1.8194
	$ (P:R)					
	(i)	145288:0				
	(ii)	145288:0				
	(iii)	0:0				

Whilst the robust sustainability ratio obtained using a mixture of flexible and non-flexible offsets was < 1 in every scenario, the nominal and opportunity ratios both remained > 1 at all times (Table 8.2). The opportunity ratio increased with time in line with increasing uncertainty in feature responses, whereas the nominal ratio began to decrease again between 50 – 100 years in response to the underlying trends assumed in the scenario (Fig. 8.2). The nominal ratio was projected to eventually reduce towards 1 over long enough timescales, as the activities carried out under offset projects became insignificant compared to change in the landscape caused by other drivers such as climate change.

Figure 8.2*: Plot of weak sustainability ratio against time, i.e. when evaluated over increasingly large timescales. Points plotted are the nominal sustainability ratio when evaluated at different points in time, where year 0 is the start of the offset policy. Uncertainty intervals correspond to the difference between the nominal ratio and the opportunity (upper limit) and robust (lower limit) ratios. Light grey line marks a ratio = 1, i.e. no net loss.*



8.3.2 *Where flexible offsets might be implemented*

Under the spatially unconstrained restoration approach, in which areas of vegetation would be restored (scenario II), the offsets resulting from existing oil and gas infrastructure would be spread out across the Ustyurt plateau, at the more central latitudes. Under the spatially unconstrained saiga habitat protection approach (scenario IV), the required offsets would be primarily concentrated into the northeast of the Uzbek Ustyurt (Fig. 8.3). In both cases these results reflect the underlying spatial distributions of the features concerned.

Applying spatial constraints to the offsets results in the prioritization of offset activities near infrastructure, as would be expected from the experimental setup. Aside from this, it does not materially alter the optimum distribution of vegetation offsets (scenario I). However, there is a slight difference under the more flexible saiga protection approach (scenario III; Fig. 8.4). Conservation priorities would change little under a no development (V) or development only (VI) scenario, which might be expected as the development footprint is such a small fraction of the Ustyurt (Chapter 6), although it should be noted that in such cases there would have been no funding for conservation generated.

***Figure 8.3**: Flexibility in type. Map showing Zonation outputs (transparent white layer) overlaid on schematic map of the Ustyurt plateau (Chapter 5). Green = priority areas for vegetation restoration, if offsetting with a multiplier of 10 and no spatial constraints. Red = priority areas for saiga habitat protection, if offsetting with a multiplier of 10 and no spatial constraints. Darker colours relate to higher importance.*

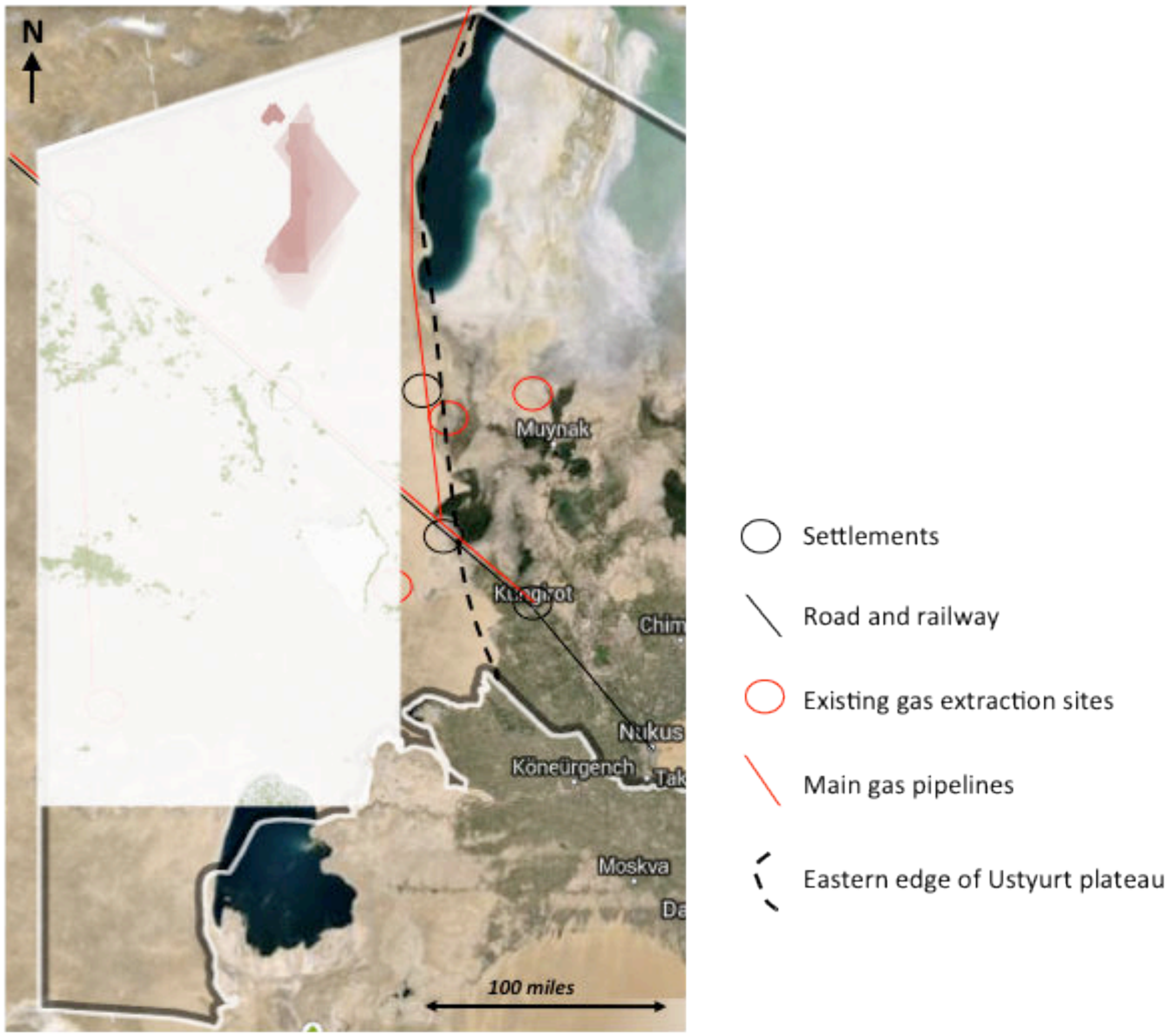
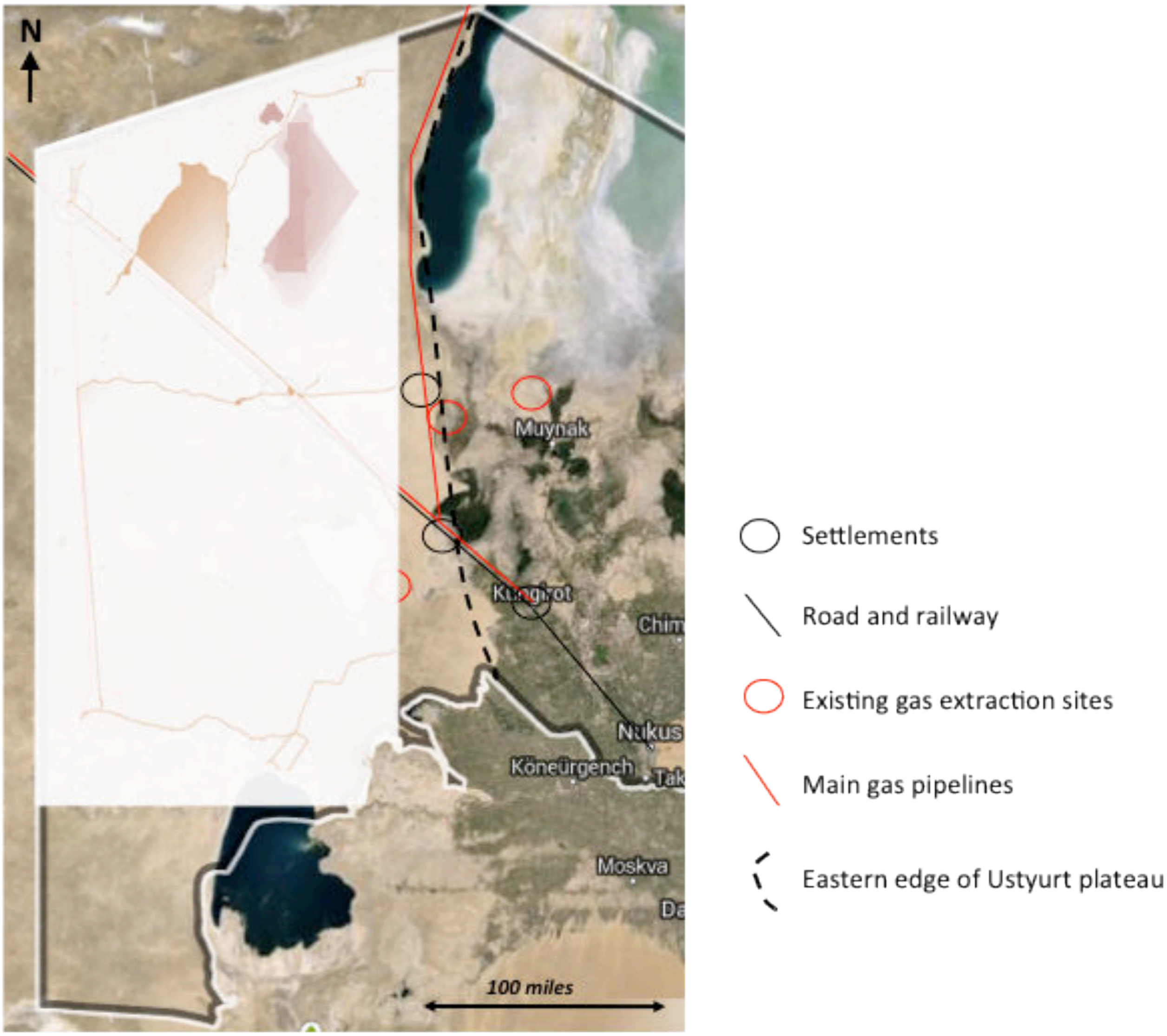


Figure 8.4: *Flexibility in space. Map showing Zonation outputs (transparent white layer) overlaid on schematic map of the Ustyurt plateau (Chapter 5). Orange = saiga offsets, if offsetting with a multiplier of 10 and spatial constraints. Red = saiga offsets, if offsetting with a multiplier of 10 and no spatial constraints. Darker colours relate to higher importance.*



8.3.3 *Sensitivity analysis*

The outcomes of the sensitivity analysis suggest that, of the RobOff input parameters varied, sustainability ratio was most sensitive to changes in the discount rate and the cost per unit area of saiga protection activities. Outcomes were robust to changes in area impacted by industry, and cost per unit area of vegetation restoration (Table 8.3). It is noted that variations in the total available budget resulted in constant sustainability ratios to a threshold of approximately $200,000, below which a decreasing ratio was obtained.

Table 8.3*: Outcomes of sensitivity analysis, with nominal sustainability ratio as the response variable. For the named parameters, the estimated value in RobOff is given alongside the range tested for that parameter. The p-value obtained for each parameter is noted, with starred significance defined against the following values: 0 (***), 0.001 (**), 0.01 (*), 0.05 – 1 (ns).*

Parameters	Estimated value	Range	P value	Significance
Total budget available ($)	800,000	10,000 – 1,000,000	0.1996	ns
Cost per unit area saiga protection ($/km^2)	2.95	0.5 – 100	0.0350	*
Cost per unit area vegetation restoration ($/km^2)	1.00	0.0001 – 10	0.5897	ns
Area impacted by industry (km^2)	500	1000 – 50,000	0.5664	ns
Discount rate (%)	0.0	0.2 – 100.0	0.0004	***

The sensitivity found to discount rate is speculated to be due to the fact that a faster response is assumed in one biodiversity feature than the other (Fig. 8.1), and changing the discount rate effectively weights earlier or later sections of these response curves.

8.4 Discussion

8.4.1 *Allowing flexible offsets*

In terms of achieving the best outcome for biodiversity conservation, these results provide support for considering the use of flexible offsets to some extent in the Ustyurt. This conforms to the limited existing discussion on offset flexibility in the literature (Overton et al., 2012; Habib et al., 2013). Under the assumptions made, RobOff selected some out of kind offsets (i.e. protection of saiga habitat) in every scenario (Table 8.2). Of course, the inclusion of a flexible offset option at all in the experimental set-up implies that the characteristic of flexibility is considered acceptable to stakeholders – in fact, whether to make flexibility an option at all is a value judgment that must be taken by policymakers. But as stated in the Introduction, the use of flexible offsetting in the Ustyurt case has some intuitive logic as funding for saiga conservation could result in more urgent biodiversity gains for the region than vegetation restoration. Equally, it can readily be seen that a flexible offset related to the protection of large core areas of saiga habitat, which could be moved in line with shifting saiga range over time, would provide a means for testing the kind of dynamic conservation solution proposed in Chapter 3.

The analyses allow deeper speculation as to whether an offsetting policy is preferable over other options at all, regardless of flexibility. In a ‘development only’ scenario, there would be 220 ± 19 km^2 of vegetation cleared, which is the current situation on the ground. Assuming that the existence of an offsetting policy did not inadvertently result in additional biodiversity components being impacted by industry, which can be a concern (Walker et al., 2009), then any offset activities at all would result in at least slightly better outcomes than development only (cf Chapter 4). In the case in which development had been prevented, however, this loss would not have occurred, but other losses would have happened nonetheless as a result of different drivers of change (e.g. the drying of the Aral sea, poaching of charismatic fauna other than saigas, and influence of other economic sectors such as agriculture or mining). Further, there would have been an opportunity cost to the local and national economy in terms of oil and gas revenue and job creation. In the no development case, had offsetting been fully implemented and feature responses been better than predicted, offsetting would also have outperformed ‘no development’ scenario from a narrow, Ustyurt-centric conservation point of view (Fig. 8.2). This conforms to the expectation laid out in Chapter 4 that offsets can outperform ‘no development’ scenarios in landscapes that are deteriorating in the absence of offsets. But the results here suggest a note of caution – when allowing for uncertainty in the responses of biodiversity features to management, this expectation might not always be met.

8.4.2 *Choosing the location of offset sites*

If implementing saiga protection activities as part of an offset scheme, the Zonation analyses suggest that all activities should be concentrated in the far north east of the Uzbek Ustyurt, near the remnant Aral Sea (Fig. 8.3). This is not a trivial conclusion. It is a conclusion that has perhaps been reached because this is the region in which the greatest numbers of saigas have been observed, and the fact that Zonation treats connectivity between sites as an important factor – so, even though saigas have in fact been recorded throughout the plateau (cf Chapter 5), this core area in the north east is prioritized. The finding supports the option of using an offset mechanism to strategically help create a new ‘Saigachy’ reserve, as this is approximately the same location as that which would be set aside under such a plan (Esipov et al., 2009).

In implementing saiga protection offsets, there is an argument for also allowing flexibility in the spatial location of offsets. Requiring saiga offsets to be close to development (i.e. non-flexible offsets) would spread saiga protection measures out across the plateau and shift the core area of protection westwards (Fig. 8.4) – on the whole, this would have the effect of protecting areas less suitable for saiga populations, and create a protection network that was probably more difficult and expensive to manage logistically. More prosaically, and even though a common principle of offsets is that they are as close to the corresponding development sites as possible (e.g. Defra, 2009), saigas are known to avoid areas of human infrastructure (Singh et al., 2010a). Therefore, protecting them near infrastructure would not necessarily be logical. Conversely, offsets involving vegetation restoration were determined by RobOff to be most suitable in cleared environments (Table 8.2) – almost certainly due to the potential gains assumed possible for vegetation, which are very large in comparison to potential gains for saigas in this environment – so for these activities spatially constrained offsets would be appropriate.

8.4.3 *Combining flexible and non-flexible offsets*

The outcomes from the RobOff analyses demonstrate that a combination of flexible and non-flexible offsets can provide the best conservation outcomes, rather than one strategy or another. The fact that the optimal proportion of budget spent on each type of offset did not remain constant across different timescales (Table 8.2) highlights the well-established point that consideration of long-term trends and counterfactuals should influence the design of conservation interventions (e.g. Willis & Birks, 2006; Poiani et al., 2010) In addition, the ongoing decrease in the strong sustainability ratio over time (Table 8.2) suggests that care is required to ensure that flexible offsetting does not result in very bad conservation outcomes for any one biodiversity feature over time. A useful piece of further research would be an exploration of the extent to which allowing trade between a mixture of biodiversity components, that are treated as interchangeable, might unexpectedly result in very poor outcomes for some of those components.

In this chapter, I have examined two types of flexibility: flexibility in the type of biodiversity that is lost and gained, and flexibility in how close the offset must be to development sites for which it compensates. Whilst I have extended the discussion around these categories of flexibility, I have stopped short of attempting to detail a comprehensive framework for the various types of flexibility in offsetting. Other categories of flexibility do exist, which I touch upon in Chapter 9: e.g. the mechanism of mitigation banking (Bekessy et al., 2010) can be effectively considered an offsetting mechanism that allows flexibility in time. Other categories might include one relating to land tenure: i.e. there is a degree of flexibility as to whether offsets are implemented on public or private land (Gordon et al., 2011b), although this is not particularly relevant for the Uzbek case study (Robinson et al., 2012).

8.4.4 *Study limitations*

The study reported upon in this Chapter is subject to a number of important limitations. Foremost amongst these is that data sets used are relatively poor or are imperfect proxies, to the extent that the analyses are robbed of sufficient predictive power to be used for detailed policy development. Really, the main conclusion that can be drawn from the practical decision-maker's point of view, in the Ustyurt, is that flexible offsetting has some merit as an option. It would be interesting to now carry out the analytical approach reported here in relation to a data-rich case study, and explore to what extent this conclusion holds.

Further, and as a result of the above, the uncertainty envelopes used in feature responses are so large that they almost certainly overwhelm the results. The methods used here could be repeated so as to experiment with a range of different uncertainty envelopes and feature response curves, allowing an exploration of the model parameter space. However, this would represent another chapter in itself, and is consequently beyond the scope of this thesis.

Finally, Zonation and in particular RobOff were developed relatively recently, and there were consequently only few examples of their application in the literature to draw upon. Whilst this study captures an attempt to use both, it is certainly the case that neither programme was implemented to its full potential. As the experimental basis grows for the use of these software packages, understanding about appropriate analyses will become more refined.

8.4.5 *Further considerations*

The robust sustainability ratios reported in Table 8.2 were < 1. But nominal ratios – those most likely to be considered by decision makers, who may be more interested in predicted outcomes than uncertainty envelopes – were generally > 1 (i.e. no net loss would be achieved). Those who implement a real world offset policy and consider only the nominal case, i.e. with a lack of consideration for uncertainties, thus risk erroneously considering no net loss to be achievable, which should be accounted for by scientists working in offset design. Further, RobOff does not allow the separate consideration of practical challenges such a lack of compliance in implementation, which is a source of uncertainty that can have a very large influence on biodiversity outcomes (Chapters 2, 4, 7). Implementation issues were included here only by proxy, in the feature responses (Fig. 8.1). Therefore, if a nominal sustainability ratio of slightly greater than 1 is predicted to be achievable in theory, it is possible that no net loss would not be achieved in practice as a result of implementation issues being more severe than expected.

It is concluded that allowing flexible offsets shows potential for improving conservation outcomes over only allowing in-kind offsets in the case of the Ustyurt. This also suggests that allowing flexibility in offsetting more generally could be useful once counterfactuals and moving conservation targets are considered. To finish on the case study in particular, based upon the analyses here and throughout this thesis, a strong case could be made in the Ustyurt for implementing *both* vegetation restoration offsets near development sites, and saiga habitat protection offsets. But this should not be seen as the limit to offsetting in this region. Although vegetation and saigas were identified as key conservation targets for the region (Chapter 5), there are other biodiversity features that should be considered in the design of offsets for the plateau, including a variety of national Red List bird, mammal, reptile and plant species, and a vast assemblage of invertebrate and microfauna species about which little is known (Chapter 5). However, given a carefully designed methodology, consideration of the issues raised in this thesis, and compliant and closely monitored implementation, it seems feasible that biodiversity offsets have the potential to achieve no net loss in this part of the world.

Chapter 9

Discussion

"Ultimate excellence lies
Not in winning
Every battle
But in defeating the enemy
Without ever fighting."

Sun Tzu (from "The Art of War", 5th Century BC)

Biodiversity offsets are an increasingly important component of the biodiversity conservation toolkit, and continue to proliferate worldwide. The debate rages over whether offsetting can ever meaningfully achieve no net loss of biodiversity in practice, although many who are doubtful would still concede that the approach presents opportunities (e.g. Pickett et al., 2013; Brown et al., 2014; Curran et al., 2014). But the science of offsetting has yet to fully catch up with the practice. It is crucial that scientists continue to develop more rigorous theory as to how offsets can be implemented in a way that most effectively meets conservation objectives. Whilst the acceptability of using offsets in the first place requires a value judgment to be made, once it has been decided that they are an option, science can inform practitioners and decision makers when and where to apply them. At this stage, a set of fundamental research themes – that require further exploration if practitioners are ever to make offsetting successful in achieving its objectives – is now well established in the literature (e.g. Gibbons & Lindenmayer, 2007; Maron et al., 2012; Gardner et al., 2013; Chapter 2). The ongoing research effort into these disparate themes is resulting in findings that will allow offsets to be better designed and implemented, and at the same time, is uncovering new insights for conservation science in general.

Of those numerous themes that have been highlighted in the literature (Gardner et al., 2013; Chapter 2), there are three for which a particularly urgent research need has been implied or explicitly suggested, and upon which I have focused in this thesis. They relate specifically to the moving target problem in contemporary conservation (Chapter 3). One is around how to demonstrate the additionality of biodiversity gains from offset projects, and thus ensure no net loss, which in particular requires a consideration of baselines and counterfactuals (Gordon et al., 2011a; Gardner et al., 2013; Maron et al., 2013). A second theme is how to determine biodiversity losses associated with economic development, and convert these into requisite gains from offset projects (Quétier & Lavorel, 2012; Gardner et al., 2013). Thirdly, a question remains over whether out of kind offsets are ever appropriate, and where they might be effectively used (Quétier & Lavorel, 2012; Gardner et al., 2013; Habib et al., 2013). This thesis has contributed to better understanding in these three areas, and I discuss each of these in turn here.

Subsequently, I briefly discuss another fundamental question, which remains open. That is, how effective offsets have been in practice to date. Whilst there have been a few empirical studies of projects in North America (e.g. Quigley & Harper, 2006; Matthews & Endress, 2008), and some studies on proxy measures such as planning permissions granted (Gibbons, 2010; Regnery et al., 2013) and restoration success (Curran et al., 2014), no one is yet able to say with any authority whether offsets work on the ground. This is largely as a result of post-implementation data either not being available in one place, or not being in the public domain at all.

9.1 Three key research themes, recommendations, and further research

9.1.1 *Baselines, counterfactuals and the frame of reference*

It has been proposed in the literature that the definition of a baseline and counterfactual for conservation interventions is a pre-requisite for evaluating true conservation progress (Ferraro & Pattanayak, 2006; McDonald-Madden et al., 2009). This has also specifically been stated for biodiversity offsetting (Gordon et al., 2011a), and is one way of partly ensuring that offsets are designed giving due consideration to moving targets. Here, I have built upon this basis and used simulation models to explore the different types of baseline or counterfactual that could be specified for an offset project. The resulting framework brings in consideration of: the background trajectory of the ecosystem in question, the ongoing management practice applied to that ecosystem, the spatial scale upon which a conservation intervention is to be evaluated, and the point in time at which the intervention is evaluated (Chapter 4). As was expected, the choice of baseline or counterfactual used in simulations determined the extent to which no net loss could be achieved in principle. As a novel component to the discussion, it was suggested that the choice of baseline or counterfactual could also influence the expectations of stakeholders in a landscape, and the incentives facing landscape managers undertaking offset projects.

An additional finding was that, if a counterfactual reflecting the status quo (i.e. development but no offsets) was used to evaluate progress, then no net loss could only ever be achieved in ecosystems that were experiencing ongoing background deterioration (Chapter 4). An example of such deterioration might be the spread of already introduced invasive species (e.g. Gordon et al., 2011a), or habitat degradation as a result of climate change. Given that this counterfactual is an intuitively sensible one for calculating net outcomes in a dynamic ecosystem, this finding should be considered in determining the applicability of offsetting to a given region – although in practice, many ecosystems are currently in decline in any case (Butchart et al., 2010). At the same time, however, this issue is problematic – it is not straightforward to project trends in biodiversity, and if offsets involve developers estimating future declines, there may be a perverse incentive to overestimate declines. The topic of perverse incentives in offsetting deserves further research.

In designing or evaluating conservation interventions, it is known to be important to consider not only the baseline and counterfactual, but also the historical context (Pooley, 2013) and interactions between drivers of change (Nicholson et al., 2009). These can together be called the 'frame of reference' for conservation interventions (Chapter 5). Whilst there are examples of the retrospective development of a counterfactual to evaluate recent interventions (e.g. Andam et al., 2008), and the need for projected counterfactuals for intervention design has been discussed (Maron et al., 2013), there are no examples of the latter in the literature. An attempt was made here to develop a basic frame of reference, including projected counterfactuals, for the Uzbek case study (Chapter 5). Whilst the analyses undertaken relied on limited data and were subject to large uncertainties, and the interactions between drivers of change are only partly understood, it was found that a useful frame of reference could be developed. The outputs demonstrate that a paucity of data should not be a reason to ignore the need for a frame of reference, although it is sometimes suggested to be (TEEB, 2010). Nonetheless, counterfactuals are not only subject to uncertainties, but qualitatively different counterfactuals can also be projected based upon the same basic trend data. To account for this, the development of frames of reference should result in the elucidation of specific questions that can be answered during ongoing implementation, allowing practitioners to modify offset schemes in a manner similar to adaptive management – whilst maintaining a commitment to no net loss of biodiversity overall.

9.1.2 *Determining the amount of compensation required*

A fundamental requirement of offsetting schemes, and one that sets them apart from other forms of compensation, is that a defensible and methodical approach is taken to quantifying biodiversity losses and gains so they can be demonstrably compared (BBOP, 2012; Chapter 2). The quantification of losses and gains is complicated when considering change in ecosystems with time (e.g. Bekessy et al., 2010; Overton et al., 2012). On the losses side, the body of literature on ecological impact assessment and the impacts of infrastructure on biodiversity is extensive. However, the Uzbek environmental impact assessment process does not currently explicitly require evaluation of development impacts upon biodiversity, and as such there is no record of biodiversity losses associated with oil and gas extraction in the Ustyurt (UNDP, 2010a). An empirical assessment of the habitat impacts caused by existing oil and gas infrastructure in the plateau was consequently required by this research project. The spatial extent of direct vegetation damage as a result of industrial infrastructure in the Ustyurt is actually rather small as a proportion of the plateau, < 1% by area, although disturbance impacts upon fauna habitat are likely to be larger (Chapter 6). This last point means that gains in vegetation are likely to be of relatively low conservation value, whereas gains in secure habitat for threatened fauna are a far higher priority. Offsets, however, do not need to be limited to one biodiversity metric (Chapter 2), and developers in the Ustyurt could reasonably be expected to compensate for both vegetation impacts and faunal habitat impacts. The main reason for limiting the number of metrics used would be to prevent the offset mechanism from becoming unwieldy and too difficult to monitor or manage.

Much effort has been spent developing various biodiversity metrics that can be used in loss-gain accounting for offsets. Each metric ostensibly has an objective related to achieving no net loss of biodiversity, but all measure different elements of biodiversity rather than the biodiversity of a region in its totality (McKenney & Kiesecker, 2010; Quétier & Lavorel, 2012). To date, there had been no case in which a set of various different offset methodologies with similar objectives had all been applied to a common case study, to test for convergence. In fact, when a range of offset methodologies is retrospectively applied to extractive activity in the Ustyurt case study, the trajectory of net gains over time resulting from different offsetting approaches diverges dramatically (Chapter 7). This is perhaps unsurprising, but has a number of implications. Firstly, the simple message communicated by offset policies to non-specialists (i.e. that they achieve no net loss of nature alongside development) is inaccurate, as on the whole it is specific sub-components of total biodiversity that are targeted. Second, methodologies are tailored to suit different national or regional conservation philosophies and institutions as well as ecological targets. This is appropriate (Chapter 5), but means that methodologies are not necessarily transferable between regions – an important point either for policymakers looking at adopting approaches to offsetting, and for multinational organizations implementing offsets in multiple regions simultaneously. Thirdly, this latter point also implies that exchange rules for trading biodiversity offset credits internationally would be necessarily and perhaps prohibitively complex.

9.1.3 *Judging how much flexibility is allowed in offsets*

Demonstrating comparability in biodiversity losses and gains associated with offset schemes is key (Gardner et al., 2013). In fact, many offsetting policies do not allow trades in biodiversity losses and gains that are out of kind, i.e. such that losses in one category of biodiversity is exchanged for gains in another (Quétier & Lavorel, 2012). However, recent research has demonstrated that using highly out of kind or ‘flexible’ offsets can prove a more effective use of conservation funding than strictly in kind offsets (Wilcox & Donlan, 2007; Brownlie & Botha, 2009; Habib et al., 2013). Nonetheless, the proposed use of flexible offsets can become highly controversial (Žydelis et al., 2009).

Research into the use of offsets in the Uzbek plateau supports the idea that consideration should be given to flexible offsetting (Chapter 2, Chapter 7). In the context of this case study, flexible offsets were taken to involve exchanging losses in the suitability of general fauna habitat in one part of the plateau for gains in suitability of saiga habitat in another. Building upon static analyses of flexible offsetting (Habib et al., 2013), I show that flexibility is also a potentially useful characteristic when considering trends in conservation targets, and to allow strategic offset networks to be designed in anticipation of future change. In addition, I suggest that a mixture of non-flexible and flexible offsets might be most appropriate for the Ustyurt (Chapter 8).

A framework has yet to be created that comprehensively captures every type of flexibility that could arise in offset schemes. Biodiversity offsets are labeled flexible if they involve out-of-kind trades in biodiversity losses and gains, e.g. they result in the loss of one type of habitat and a comparable gain in a different habitat. But these can be considered one category of flexibility, e.g. flexibility in ‘type’. It is not difficult to conceive of other ways in which offsets could be considered flexible. For instance, it is common policy to require that offsets are implemented as close as possible to the development site for which they compensate (e.g. Defra, 2011) – but in practice, this is not always possible, and so offsets take place further afield. This represents flexibility in space. If offsets have the potential to be flexible in space, then it perhaps follows that they can also be flexible in time. Indeed, this is something that already takes place within existing offset policies. If a development occurs and is then compensated for with an offset, it can take time for the ecological benefits from the offset to accrue, so there is a time lag between development losses and gains. This is often dealt with after development through multipliers (Moilanen et al., 2009). It has been argued that time lags in biodiversity gains from offsets should necessitate the use of conservation banking mechanisms, such that biodiversity gains are achieved in advance of development losses (Bekessy et al., 2010). But conservation priorities can change with time, so the implementation of an offset in advance of the development impacts for which it compensates may target different priorities than one implemented simultaneously with development. Banking can therefore be generalized as a flexible offsetting approach in time.

So, to the concept of offsets that are flexible in type, I can add offsets that are flexible in space (e.g. not being required to be on or near development sites) and flexible in time (e.g. being put in place before development occurs, through a banking mechanism). Flexibility could feasibly enter offset schemes in other ways that do not fit into the categorization of space, time or type. One example concerns allowing different land tenure arrangements upon impact sites and offset projects, (e.g. offsets could be implemented on public land with the losses occurring on private land; Gordon et al., 2011b). This might represent flexibility in terms of who benefits from the offset scheme, with not inconsiderable implications: for instance, in the example given above, the biodiversity would effectively be turned from a private good into a public one. The case study used in this thesis concerns land that is almost entirely publically owned (Robinson et al., 2013), and the implications for offsetting of allowing flexibility in land rights are therefore not relevant in this case. However, there is much work to be done building on the limited existing basis (Gordon et al., 2011b), to better understand the role of land tenure in flexible offsets. Further, and more generally, a comprehensive categorization of flexibility in offsets would be useful for decision makers, and also prove an interesting avenue for further research.

The issue of whether to allow flexibility links not only to conservation value, but also to the social objective of the study, which I have not explored here. For instance, a common requirement of conservation interventions might be ensuring human access to nature. Such an objective might provide a different argument for requiring spatially constrained (non-flexible) offsets, if that means that offset locations are closer to transport infrastructure or urban centres. This might be less of a concern for my case study region as, despite the existence of some industrial infrastructure, the settlements on the plateau are effectively surrounded by an extensive wilderness. However, it is a point that should not be overlooked in the more general application of flexible offsets.

9.2 Biodiversity offsets on the ground

Biodiversity offsets have been successful in becoming an established policy tool for biodiversity conservation, in both a regulatory and voluntary capacity, as evidenced by their public, private and third sector take-up worldwide (Madsen et al., 2011; IUCN-ICMM, 2013). Further, despite remaining controversial for many both professionally and personally (Walker et al., 2009; Blackhurst, 2012), there are numerous researchers who are currently convinced that offsetting can work in principle. In both of these respects, offsets have already been rather effective.

However, it will never be known how much can be achieved until it becomes clearer whether offsets have delivered in practice. Despite the scattering of analyses that I mention above, there has been no general evaluation of offset projects implemented on the ground (Chapter 2). In fact, this is the case for a number of relatively novel conservation mechanisms, including offsets, which could potentially be useful in conserving moving targets (Chapter 3). There are some promising signs: Gibbons (2010) explains that approvals for vegetation clearances in New South Wales fell dramatically after the introduction of an offset policy, whilst Regnery et al. (2013) find that the offsets approved by French planning authorities should result in improved species conservation outcomes compared to business as usual. Although Curran et al. (2014) demonstrate concerns about the restorability of global old growth habitat in practice, they concede that offsets could be effective for habitats with shorter maturation times. These studies provide useful context, but all assess proxy measures of offset success, rather than the actual outcomes of offset projects.

In order to carry out such an assessment, there is a need for accessible data on offset projects. Whilst some such data do already exist in the public domain, especially those arising through regulatory offset policies, records are poorly kept and difficult to collate. A key step towards better understanding of offset implementation through data analyses will be for local and national authorities to keep comprehensive and publically available registers of those offsets implemented within their jurisdiction. An example of a region that has taken a step in this direction is the US, via the Wetland Banking ‘Regulatory In lieu fee and Bank Information Tracking System’ (RIBITS).

Further, there are an unknown but potentially extensive number of offset projects undertaken on a voluntary basis by developers, as a result of corporate sustainability commitments, or in response to co-financing lender requirements (e.g. IFC, 2012). Whilst some already do, these private sector organizations could considerably improve the science and practice of offsetting by compiling data on voluntary offset implementation and making it available to researchers through a central repository of some kind. A co-benefit of such an undertaking would be to emphasize that industry does not necessarily need to be seen as the enemy of biodiversity conservation. Rather, industry should be seen as a necessary part of the socio-ecological system that could potentially be managed in such a way so as to support conservation (hence the quote from Sun Tzu which leads into this Chapter).

9.3 Biodiversity offsets in the Ustyurt

One outcome of the research captured in this thesis is the provision of concrete recommendations for the implementation of a biodiversity offset scheme in the Ustyurt plateau. The recommendations are intended to feed into the project being implemented by the UNDP (UNDP, 2010a), which has as one output the piloting of a biodiversity offset methodology for the oil and gas sector in the Ustyurt.

Overall, the findings contained within this thesis support the trial use of biodiversity offsets in the Uzbek Ustyurt. Given the recent historical context, biodiversity baseline and counterfactuals developed here (Chapters 4, 5), the focal conservation targets are suggested to be both vegetation cover and the saiga antelope population. In order to attempt and achieve no net loss, those exploring for or extracting natural gas in the region would need to, as a minimum, (a) re-vegetate and restore an area of land corresponding to their direct footprint, and (b) protect an area of land that is some multiple of their direct footprint from poaching and other activities that threaten faunal species (Chapters 6, 7). To maintain no net loss against projected regional counterfactuals, restored vegetation would need to be maintained in a stable state, whereas the saiga antelope population would need to increase to some extent (Chapter 5). Having said this, the relative lack of accessible data to project trends in the region, in comparison to other countries with offset policies, and the fact that the plateau had undergone such dramatic change in recent years due to the drying of the Aral Sea, means that close monitoring will be required to analyse and update offsetting practice based on the actual trends that do occur.

As a programme involving flexible offsets may represent the most efficient use of conservation funds in the region (Chapter 8), restoration would not necessarily have to be located close to development sites, and protection could be achieved by paying into a fund for anti-poaching activities in a protected area such as the proposed *Saigachy* reserve. However, anti-poaching activities should not be limited to this reserve, but should instead be designed to follow the saigas around the landscape, in line with the recommendations first made in Chapter 3. Perhaps one of the most unusual elements of implementing a biodiversity offset policy here, compared to other countries, is the complete lack of private land ownership: which, although limiting the scope to implement a mitigation banking mechanism (Bekessy et al., 2010), does mean that there is relative freedom in allowing saiga conservation activities to be implemented wherever necessary on the plateau.

Ensuring compliance will be an important challenge in the Ustyurt, as it is everywhere that offsets are implemented (Chapter 2), but there already exists a legal framework into which the no net loss principle could be incorporated, and within which a detailed biodiversity offset methodology could be specified. Part of this would almost certainly involve bringing biodiversity as a mandatory topic into the environmental impact assessment process. One additional means for ensuring compliance might be to promote the return of independent environmental NGOs, who might observe progress in implementing biodiversity offset schemes (Chapter 5). The legal and policy framework within which this regional offset scheme will be implemented is complicated by legal considerations relating to the proximity of the international border with Kazakhstan. Along with the Memorandum of Understanding drawn up between the two countries in relation to transboundary saiga conservation (Chapter 5), a coordinated offset policy with oil and gas companies on both sides of the border could potentially provide another opportunity to increase bilateral cooperation on biodiversity conservation.

Finally, a systematic monitoring programme is required for biodiversity losses and gains, associated with development and offset schemes. Ideally, this would produce a register of developments and associated offsets with publically available data, but whether this will become a feasible policy option in Uzbekistan is unclear. Whilst these and other recommendations arising in relation to the implementation of a biodiversity offset methodology in the Ustyurt plateau are made throughout this thesis, the headline points are summarized for ease of reference in the Appendices (Table A4.1).

Throughout, I have primarily considered the implementation of biodiversity offsets in the Uzbek portion of the Ustyurt only, although the plateau is contiguous between Uzbekistan and Kazakhstan. A relevant consideration for offsetting in the Ustyurt, and a topic that arises briefly in Chapter 7 and above, is whether international trade in biodiversity losses and gains through offset schemes should be allowed or encouraged. Whether to allow or encourage an option for international trade in biodiversity offsets is a difficult question. On the one hand, doing so would potentially enable offsetters to use conservation funds more effectively for moving targets, such as migratory species (Chapter 3). It is also a necessary consideration in discussing marine offsets, a topic which is attracting growing interest (e.g. Dickie et al., 2013). In fact, the marine offsets proposed for bycatch species in commercial fisheries (Wilcox & Donlan, 2007) provide an interesting example of how a long distance and trans-jurisdictional offset scheme might feasibly work. On the other hand, I have shown how different approaches to biodiversity loss and gain calculation can be in different jurisdictions, which might make it difficult to compare losses and gains across borders (Chapter 7). I would argue that international or very long distance trade in biodiversity offsets should be considered, but only when it is designed to resolve a specific moving target conservation problem (e.g. achieving the maximum possible gain in a migratory target species that has been impacted by development) – and only with due consideration given to the need for clear exchange rules between different jurisdictions.

9.4 Biodiversity offsets in the wider context of conservation science

Whilst the theoretical basis for offsetting continues to be strengthened – a research effort to which this thesis contributes – it has yet to be proven whether biodiversity offsetting can actually work in practice. But there are some promising signs, and so it would be premature to rule offsetting out completely at this stage in its development. Given the judicious application of offsetting to those regions only in which biodiversity deterioration is ongoing, where biodiversity restorability is feasible, where compliance and monitoring are realistic, and where a flexible approach is acceptable to stakeholders, it could potentially prove an increasingly useful tool for biodiversity conservation. What is more, a number of the barriers to successful offsetting are not problems with the approach specifically, but are problems for conservation science and practice in general.

So, for instance, the need to make offsets more robust through improved evaluation and detailed accounting for losses and gains, both of which require a better application of counterfactuals, is something in common with any other biodiversity conservation intervention (McDonald-Madden et al., 2009). It is notable that offsets are often implemented by private sector actors, rather than by the public sector or NGOs, and it may be that the private sector's strong commercial and legal needs for detailed cost benefit accounting provide a driver for improved evaluation of offsetting, and subsequently conservation interventions in general. Here, in relation to this broader problem for conservation, I have contributed some considerations on how exactly to specify different counterfactuals, and exploration as to how the choice of counterfactual can affect ecological outcomes and the incentives facing land managers (Chapters 4, 5). This should be seen in light of the parallel need for better evaluation of conservation interventions through institutional change, particularly in terms of those funding conservation, so as to overcome existing barriers to project evaluation on the ground (Bottrill et al., 2011).

In Chapter 5, I emphasize the need to give due consideration to the historical context when attempting to achieve no net loss of biodiversity. The importance of doing so for any conservation intervention is established (Pauly, 1995; Pooley, 2013). But despite the difficulties experienced here in carrying out analyses over a historical period of 100 years, many would say that this is not remotely far back enough, as it does not allow the separation of trends from natural variability (Willis & Birks, 2006). In principal, it has been argued that conservation planners should turn to paleoecology to truly understand the most appropriate biodiversity baseline, to identify and where necessary manage novel ecosystems, and to better establish what can be considered 'natural' (Seastedt et al., 2008; Hobbs et al., 2009; Willis et al., 2010). Analyses over such long timescales are few and far between for the Uzbek case study (Melville et al., 2009), which makes developing sufficiently long-term historical baselines for the region an aspiration rather than a current possibility – as it would be for many regions. The type of analysis completed in Chapter 5 should thus be seen as one practical approach to contextualizing conservation interventions given limited data, but which should ideally act as a stepping stone to developing a more comprehensive frame of reference.

An important and perhaps surprisingly difficult question for conservation – that is brought into sharp relief by the practice of offsetting, and in particular the specification of a no net loss objective – is what it is exactly that interventions seek to conserve. Offsetting has resulted in the development of a plethora of metrics and methodologies associated with the calculation of biodiversity losses and gains, and which I have explored further here (Chapters 6, 7). Research such as this has, in turn, shone further light on wider issues such as whether for a given intervention it is an individual species that is being conserved, or an assemblage of species, or the ecological function provided by a certain type of species, or the services to human society provided by a species or assemblage, and so on. This type of consideration is more than mere semantics: for instance, if the goal of an intervention is the conservation of ecological goods and services, then the degree of social equity in the receipt of these goods and services becomes an additional question, and one which may affect the design and outcome of the intervention (Halpern et al., 2013). In general, the global achievement of biodiversity conservation requires not *more* protected areas, but *better* protected areas (Fuller et al., 2013). The latter will in part be achieved by developing an improved understanding and clarity about what biodiversity we want to retain, and why we want it – an area in which biodiversity offset research can play a supporting role.

Finally, conservation science has yet to successfully meet the challenge presented by accepting that all conservation targets are essentially moving targets on a large enough timescale. One strength of the offsetting approach is its flexibility, in that it provides a framework for designing exchange rules for biodiversity components that are different in space, time, type and so forth – a necessary tool for moving target conservation. In this thesis, I have extended previous research by exploring the applicability of flexible offsets, and how flexibility might benefit moving targets (Chapters 3, 8). The need for conservation interventions that are more flexible, and the controversy surrounding the design of them, is highlighted by recent proposals for practical conservation strategies in the US based upon suggestions made in this thesis (BenDor & Woodruff, 2014). It is in finding solutions to moving target conservation problems that biodiversity offset research has perhaps its greatest general contribution to make.

All things considered, the useful question in relation to biodiversity offsets is perhaps not: "do offsets work". Rather, it is: "offsets *should* work. What are the reasons they do not". Finding the answers to *that* question – something I hope to have contributed towards with this thesis – will not only allow improvements to be made to the practice of offsetting. It will also help us to improve conservation science.

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Appendices

1. Additional model specifications, and results obtained with different trajectories
   (Chapter 4)
   - Detailed description, generalised model
   - Detailed description, Melbourne model
   - Outcomes with different functional forms of biodiversity trajectory

2. Additional analyses and results
   (Chapter 5)

3. Additional detail on biodiversity offset methodologies used in calculations
   (Chapter 7)

4. Summary of practical recommendations for biodiversity offsetting in the Uzbek Ustyurt
   (Chapter 9)

Appendix 1

A1.1 Detailed description – generalized model

The general model was entirely written in the statistical programming language R version 2.15.1 (R Development Core Team, 2012). The most important simplifying assumptions made in building the model are as follows.

Model assumptions

The purpose of the simple model was not to simulate real-world biodiversity offsets, but to explore the principle behind offsetting and show the effect of varying the baselines used to evaluate performance. Thus a number of unrealistic simplifying assumptions were made, ignoring the majority of design challenges to offsetting unless where explicitly stated in the manuscript.

- Biodiversity trajectory is either of constant or logistic (+ve / -ve) functional form. Alternative functional forms presented in Appendix A1.3;
- Development 'project' impacts to be offset occur all at once and instantaneously;
- Development impacts are all negative and linear (constant decrease in biodiversity value every time step);
- Offsets happen instantaneously and at the same time as development 'projects', completely negating development impacts;
- Offsets are fully compliant and successfully implemented;
- No limit is placed in the ability of offsetters to create extra biodiversity value equivalent to biodiversity losses, but;
- Once an offset has been created it is managed, and managed land does not then follow any biodiversity trajectory;
- Biodiversity losses and gains are both measurable and known exactly;
- All developers and offsetters act transparently and in compliance with the offset policy; and,
- All development is offset, nothing else except development projects and biodiversity trajectory changes overall biodiversity value.

A number of these assumptions are relaxed and replaced with real-world complexity, such as time lags between development impacts and offset projects, in the predictive model (A1.2). An overview of some of the key design and implementation challenges in offsets can be found in Chapter 2.

Model parameter values

As the model is theoretical and used to examine relationships only, the model parameters used in running simulations were arbitrary and not based on any real-world parameters. The values are consequently not included in the main text, to avoid confusion. For the sake of reproducibility, they are included in Table A1.1.

Table A1.1: *Parameter values used in theoretical model simulations*

Parameter	Value	Range tested
t	n/a	0 – 100
B_0	500	n/a
$p(t)$	1	n/a
k_1	0.1	0.05 – 0.35
k_2	-5	n/a
m	1	0.5 – 4.0

A1.2 Detailed description – Melbourne model

Study area and context

Melbourne is situated amongst some of the best remnants of Western (Basalt) Plains Natural Temperate Grassland. This vegetation community has over 99.5% of its original distribution lost or substantially altered (Williams et al., 2005) and comprises one of Australia's most endangered ecosystems. The extent and condition of native grassland in the model is shown in Fig. A1.1.

The native grassland condition map used in this model was derived from a vegetation condition layer for the state of Victoria (GIS layer NV2005_QUAL1; DSE, 2009) and from site data collected during 2008-2009 by the Victorian Growth Areas Authority and the Victorian Government's Department of Sustainability and Environment (DSE). Each cell represents grassland condition within a 50 x 50 m^2 pixel with condition measured using the habitat hectares vegetation assessment method (Parkes et al., 2003) for areas where sites were visited and with a derived habitat hectares score from the modelled vegetation condition layer in other areas. The habitat hectares score ranges between 0-100 and represents vegetation condition relative to a mature and undisturbed benchmark of the same vegetation type. The score includes components that describe site condition (75% of score) and landscape context (25% of score). For this analysis, only the site condition component was used and the grassland condition map was then rescaled to range between 0.0 – 0.75 (Fig. A1.1).

Grassland condition change model

All the parameters described below were estimated based on the expert opinion of DSE ecologists. Additional references providing further background information for the condition change model are McDougall and Morgan (2005), Morgan (1998), Lunt et al. (2007) and McLaren et al. (1998).

The curves shown in Fig. A1.2 describe how future grassland condition was calculated using the *habitat hectares* metric (Parkes et al., 2003). The curves indicate that unmanaged grasslands degrade over time and that managed grasslands will improve in condition. However, once a patch of grassland falls below a certain condition, it is likely to be difficult to restore it to good condition. Thus there were managed condition curves for grassland above and below a threshold condition of 0.35 (Fig. A1.2) and when managed, grassland with a condition score above/below the threshold will asymptote towards a value of 0.75/0.35, respectively. A small proportion of pixels were allowed to cross the threshold in an upward direction due to random fluctuations and embark on the higher recoverability curve if managed. The probability of a pixel being allowed to cross the threshold was determined by expert opinion and set to 0.0005.

For each time step, the condition of each pixel of grassland was evolved using the curves in Fig. A1.2. This was done with an algorithm that operated using the following procedure on each pixel of grassland: i) The appropriate curve was chosen depending on whether the pixel of grassland was unmanaged, managed above threshold or managed below threshold; ii) The point on the curve that matched the current condition of the pixel was determined, call this (t_1,c_1); iii) The condition of the pixel for the next time step was then determined by traversing the curve a distance determined by the time-step used in the model (call this s), which in this case was $s = 100$ years. If the condition change curve was described by the function $f(t)$, then the new condition (c_2) is given by $c_2 = f(t_1 + s)$.

To model stochasticity of the condition change process, the condition score of each pixel was perturbed by adding a small random value sampled from a normal distribution with a mean of zero and a standard deviation of 0.02 (estimated from expert option and varied in the uncertainty analysis). Finally, moving window smoothing was applied to provide an edge effect. The edge effect models an environmental weed invasion front - where regions supporting lower condition grasslands or non-grassland supply weed propagules to adjacent comparatively weed free regions. Without active management, initially weed-free regions that are proximal to propagule sources will in turn become sources of weed propagules themselves. As such, the rate of decline in the condition of a given grassland area will depend on its spatial context at each time step. The moving window smoothing was implemented with 5 x 5 cell Gaussian kernel. Areas were counted as a weed source and used in the smoothing calculation if they were either non-grassland (condition score = 0) or had a condition score below 0.12 (these values were estimated from expert opinion).

Figure A1.1: *The study area west of Melbourne, Australia. The map shows grassland condition as a graduated greyscale (darker colours show higher condition) and property parcels in the study area (from Gordon et al., 2011a)*

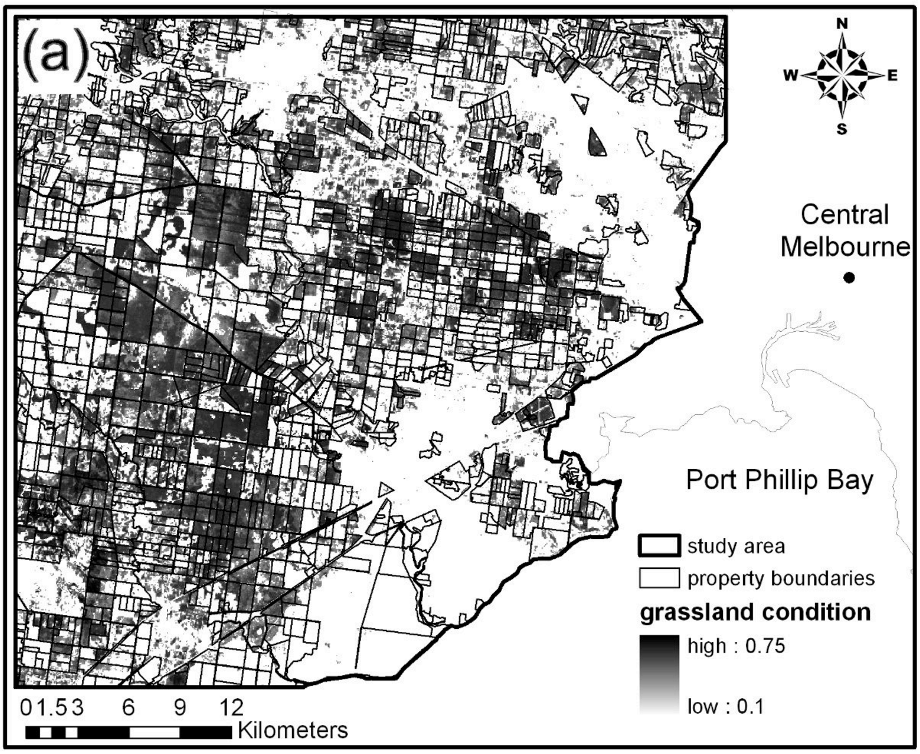
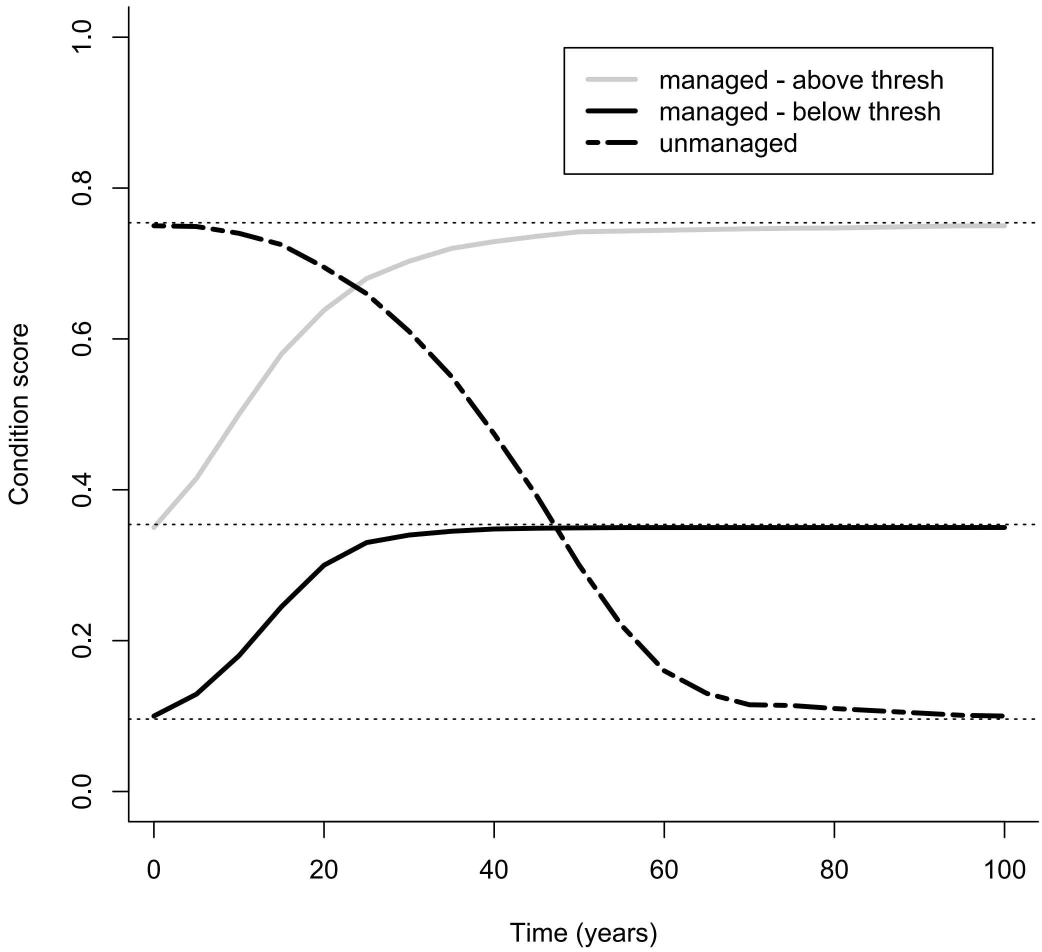


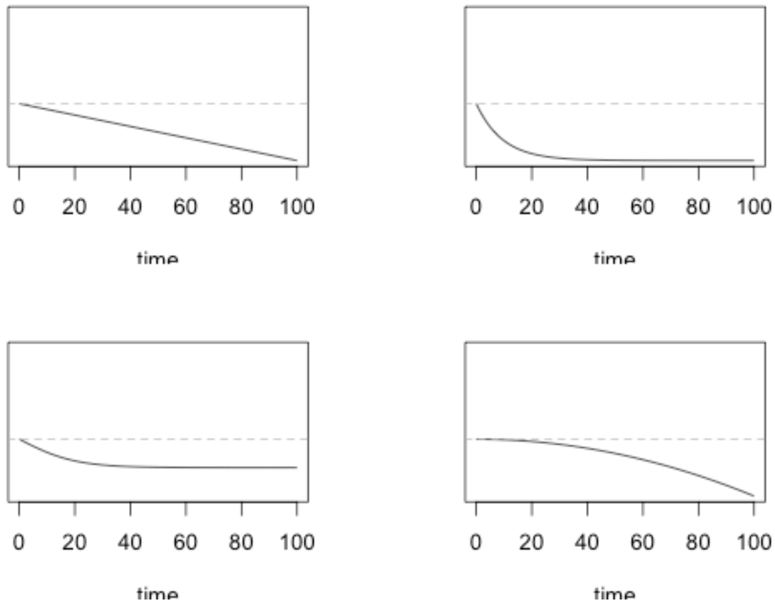
Figure A1.2: *Curves used to parameterise the grassland condition model. These curves are used to predict condition change over time for managed (solid lines) and unmanaged (dashed line) grasslands. The lower solid line represents vegetation whose initial condition is less than the 0.35 threshold value. The upper solid line represents vegetation whose initial condition is above the 0.35 threshold (from Gordon et al., 2011a)*



A1.3 Outcomes with different functional forms of biodiversity trajectory

The functional form of biodiversity trajectory used in Chapter 4, excluding the ‘stable’ trajectory, is of logistic growth or decline. This is one of a number of possible generic trajectories for conservation targets, and so we explored the effect of using different functional forms upon model outcomes. These forms were: constant, constant proportion, logistic, and increasing proportion (Fig. A1.3), and are based upon those outlined by Mace et al. (2008).

***Figure A1.3**: Different generic functional forms for biodiversity decline (reversed for biodiversity increase). Forms are (a) constant decline, (b) constant proportion decline, (c) logistic decline, and (d) increasing proportion decline. Adapted from Mace et al. (2008).*



In reality, the trajectory followed by any one conservation target might be a variant on any of these, some combination of these, or none of them. However, recent work has shown that these functional forms describe the trajectories of many Red List species very well (Di Fonzo et al., under review), and the logistic form is similar to that used by Gordon et al. (2011a) for the Melbourne model (Appendix A1.2), and is based upon empirical measurements of grassland condition change. Consequently, we feel that for the purposes of the hypothetical model, this range of functional forms (especially when taken with their positive counterparts, and the 'stable' trajectory) are adequate.

When we apply the additional three non-logistic functional forms (a, b, d in Fig. A1.3) as deteriorating biodiversity trajectories in the model, and evaluate against a fixed baseline, we obtain qualitatively comparable outcomes as for the logistic form in Chapter 4 (Fig. A1.4, A1.5, A1.6).

Figure A1.4: *Plots for comparison with Figure 1 in the main manuscript, but with the linear biodiversity trajectory (Fig. S3.1a). Plots of biodiversity value ('condition') against time under different biodiversity dynamics, using a fixed baseline (dashed grey line), on a landscape scale. No scale is given here for B(t) on the y axis, but B(t) increases away from the origin. Each plot represents a different biodiversity trajectory, with biodiversity trajectory: (a) deteriorating; (b) stable, where black and green lines coincide; and, (c) improving*

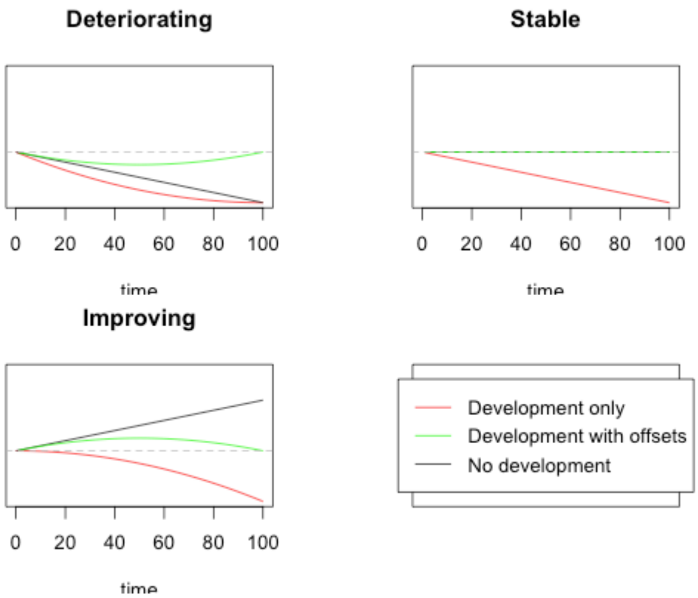


Figure A1.5: *As for Figure A1.4, but with a constant proportional change in biodiversity trajectory (Fig. A1.3b). Note shortened time span in the x axis in (c)*

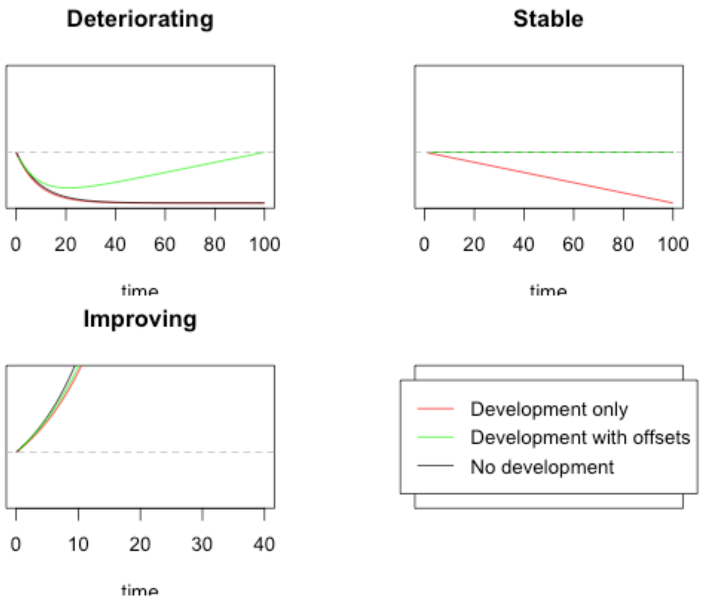
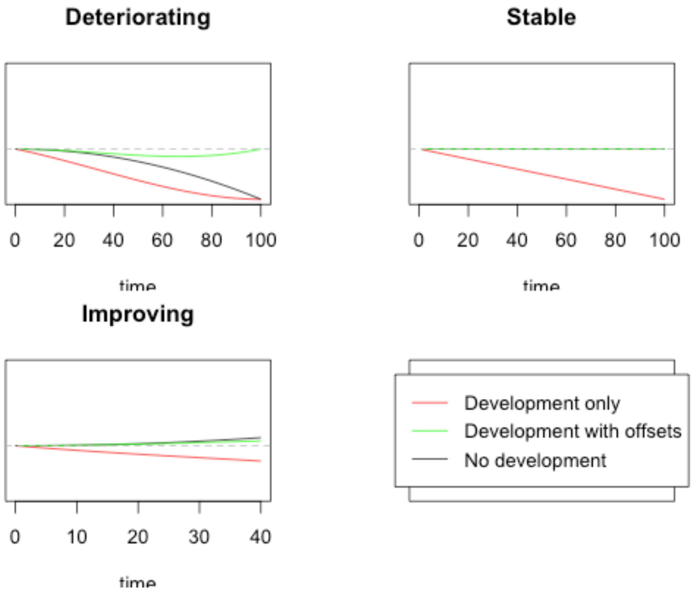
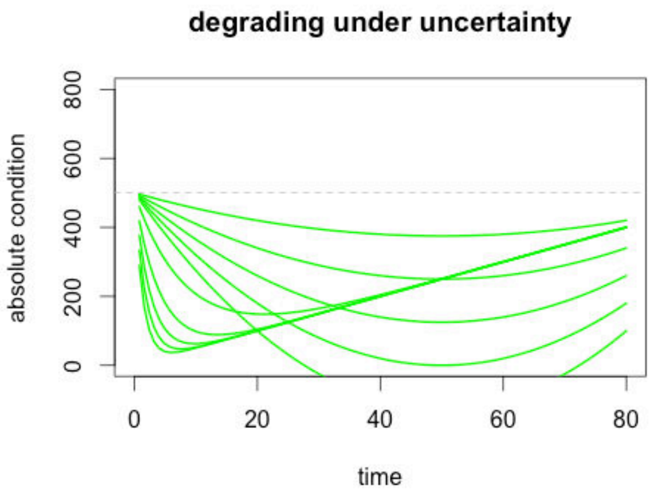


Figure A1.6*: As for Figure A1.4, but with a increasing proportional change in biodiversity trajectory (Fig. A1.3c). Note shortened time span in the x axis in (c)*



I also repeat the analysis used to explore the influence of uncertainty in both biodiversity trajectory parameters and implementation success *m*, via which the results in Chapter 4 were obtained. Under the same conditions, but with both a linear decline and constant proportional decline in biodiversity trajectory (Fig. A1.4a, b respectively), again the outcomes are not qualitatively different and the condition curves converge (Fig. A1.7).

Figure A1.7*: The outcomes of a 'development with offsets' scenario, against an absolute baseline, under two different deteriorating biodiversity condition trajectories. Subject to variation in the parameters used to specify the trajectories, as for Figure 2 in the main manuscript. Trajectory is constant (cf Fig. A1.3a) for the set of curves with minima between time = 0 and time = 20, and constant proportion (cf Fig. A1.3b) for the set of curves with minima for time > 20.*



The fact that the mode outcomes are eventually qualitatively equivalent for the range of functional forms (Fig. A1.4 – A1.6) to those found using the logistic biodiversity trajectory (Chapter 4), means that the evaluation of outcomes against counterfactuals also results in equivalent conclusions being drawn regarding the no net loss objective. Consequently, the outcomes obtained using the various different functional forms (Fig. A1.3) would result in analogous outcomes to those shown in Chapter 4.

Appendix 1: additional references

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Appendix 2

This appendix to the main text contains additional technical information on the main data sets used, as well as tables and figures resulting from certain further analyses. These were not included in the main body of the manuscript for the sake of brevity.

To begin, a summary of all data sets analysed here and in the main text (Table A2.1).

***Table A2.1**: Main data sources with regards to landscape dynamics in the Ustyurt plateau*

Area of analysis	Data set	Description	Time series	Analysis	Source
Conservation target: habitat	Normalised Difference Vegetation Index (NDVI)	NDVI data for northwestern Uzbekistan, projected in WGS84, resolution 8 km^2 (GIMMS), 1 km^2 (KARS and MODIS)	1981 – 2006	Trend in mean spring/summer NDVI	*AVHRR GIMMS data set (ftp://ftp.glcf.umiacs.umd.edu/glcf/GIMMS/Geographic/)*
			1991 – 2003	Spatial trend in spring NDVI	*KARS data set (University of Kansas)*
			2002 – 2012	Spatial trend in spring NDVI	*MODIS NDVI data set (https://lpdaac.usgs.gov/get_data/)*
Conservation target: habitat	Vegetation type	Vegetation types, distribution, and dominant species on the Ustyurt plateau	Compiled 2003 Compiled 2011		*Gintzburger et al., 2003* *Esipov & Shomurodov, 2011*
Conservation target: species	Saiga (*Saiga tatarica*) population	Ustyurt population abundance, aggregation size, sex ratio, limited poaching statistics	1980 – 2011	Trend in abundance of saiga Trend in saiga distribution	*Bekenov et al., 1998* *Bykova & Esipov, 2011*
		Participatory monitoring data	2006 - 2012	Trend in average winter latitude of saiga	*Institute of Zoology, Kazakhstan*
Driver: physical	Meteorological data (Jaslyk and Karakalpakia)	Temperature, precipitation and snow cover data.	1970 – 2010	Trend in annual mean temperature, winter temperature, annual sum precipitation.	*E. Bykova*
		Wind speed data	2009 – 2010	Dominant wind direction	*I. Aslanov*
Driver: physical	Ecological atlases	Water availability, salinisation of water courses	Compiled 1983 Compiled 2008	Trend in water availability and salinisation	*CCCP, 1983* *Akcura et al., 2008*
Driver: social	Demographics	Number of people, Karakalpakstan and Uzbekistan	1959 – 2011	Trend in number of residents	*UN, 2011*
Driver: social	Socio-economic	Reports of socioeconomic surveys	2004 2008 2011	Trend in unemployment rates	*Bykova & Esipov, 2004* *Kühl, 2008* *Phillipson & Milner-Gulland, 2011*
Driver: economic	Macro-economic trends	GDP, industrial production, transport, employment	1991 - 2010	Trend in unemployment rates	*Dynamic Lines, 2011*
Driver: economic	Agriculture	Livestock numbers and land use types, Karakalpakstan	1990 - 2010	Trend in head of livestock	*UNDP, 2010a* *JICA, 2011* *Karakalpak Ministry of Agriculture*
Driver: economic	Oil and Gas infrastructure	Location of main pipelines, map of exploration rights, location of key gas facilities and road network	Compiled 2008	Trend in quantity of oil and gas extraction Trend in distribution of oil and gas	*Yenikeyeff, 2008* *Akcura et al., 2008* *Field survey (J. Bull)* *Jones et al., 2014*
			Compiled 2012	Impacts of natural gas infrastructure on habitat	

Technical details: satellite data

There exist a number of vegetation indices based upon satellite data in common usage for conservation and land management purposes; including most commonly the Normalized Difference Vegetation Index (NDVI), but also the Global Environment Monitoring Index, the Soil Adjusted Vegetation Index, the Modified Soil-Adjusted Vegetation Index, the Transformed Soil Adjusted Vegetation Index, the Perpendicular Vegetation Index (PVI), and more (Leprieur et al., 1996; Bannari et al., 1995). Of these indices, the NDVI is the preferred choice because (a) it is the most widely used, recognized and understood (Pettorelli et al., 2005), and (b) pre-calculated and processed NDVI data sets already exist. NDVI data is calculated using an index of the difference between the Red and Near Infra-Red spectral bands.

In the case of the Ustyurt, vegetative cover is low but in certain seasons exceeds the threshold level of 20 – 30 % vegetative cover above which the NDVI is a useful measure of green biomass (Kennedy, 1989). Esipov & Shomudurov (2011) found spring ground cover to average 52 ± 19% at 19 sites across the Uzbek Ustyurt. Gintzburger et al. (2011) surveyed summer vegetation cover and found values to range between 13.73% and 62.12% of line transects completed. From approximately mid-summer onwards the dominant vegetation types (*Artemisia terrae-albae, Anabasis salsa, Salsola arbusculaformis, Halyoxlon aphyllum*) turn brown, which means they would not be captured in the NDVI (Robinson, 2000). In winter, large areas of the ground surface are obscured by snow or ice, and vegetative growth is completely suppressed by the cold temperatures (Gintzburger et al., 2003), which would make NDVI a poor indicator of vegetative cover.

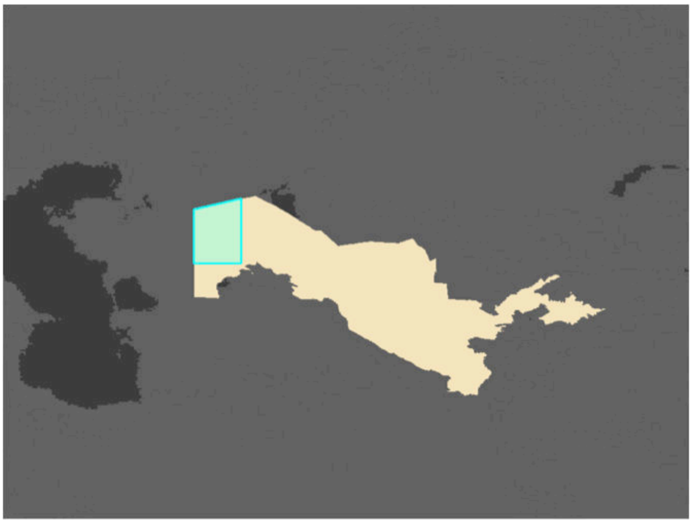
As a result of these considerations I used NDVI for the analyses here, but only during springtime, and only to examine trends (i.e. rather than to estimate actual vegetation cover on the ground). I used three different NDVI data sets – labeled GIMMS, KARS and MODIS – obtained by different satellites for various spatial and temporal scales. These datasets have been cleaned for atmospheric and positional effects (Pinzon et al., 2004).

GIMMS data set

To examine long-term trends, I used the AVHRR (Advanced Very High Resolution Radiometer) GIMMS (Global Inventory Modelling and Mapping Studies) NDVI data set, which is available for the period 1981 – 2006, at 4km resolution (Pinzon et al., 2004). Image files were taken from the Global Land Cover "ftp." server, and exported into ArcGIS. I use the GIMMS data set to explore NDVI for an area of interest approximately corresponding to the study region (Fig. A2.1). The utility was to examine initially whether (i) average NDVI showed clear trends during the study period, and (ii) whether this was strongly linked to trends in climate.

Using the 'Spatial Analyst' package, mean/sd NDVI across the area of interest (Fig. A2.1) were calculated for every 15-day increment in the period 1982 – 2006. The area of interest was defined using a simple four-sided polygon within Uzbek territory, west of the Aral Sea, and north of the border with Turkmenistan.

***Figure A2.1**: Area of interest for which mean/SD/CV NDVI values were calculated using GIMMS data set.*



KARS data set

To examine vegetation cover trends at higher resolution, the preprocessed data set produced by the Kansas Applied Remote Sensing Program, at the University of Kansas ('KARS') was acquired. This is also based on the AVHRR, and gives NDVI at 1 km resolution, spatially referenced, for all of Central Asia, for the period 1991 – 2003.

The KARS data set was used to examine the spatial distribution of any trends in spring NDVI in the Ustyurt. In order to do this, average spring NDVI was calculated by pixel, and then the separate annual spring NDVI layers were combined such that the change in NDVI was fitted to a linear model for each pixel, as described in the main text. The linear formulae were used to estimate change in NDVI by pixel from the base year (1991) during the period for which data were available, and the values were presented in raster format.

MODIS data set

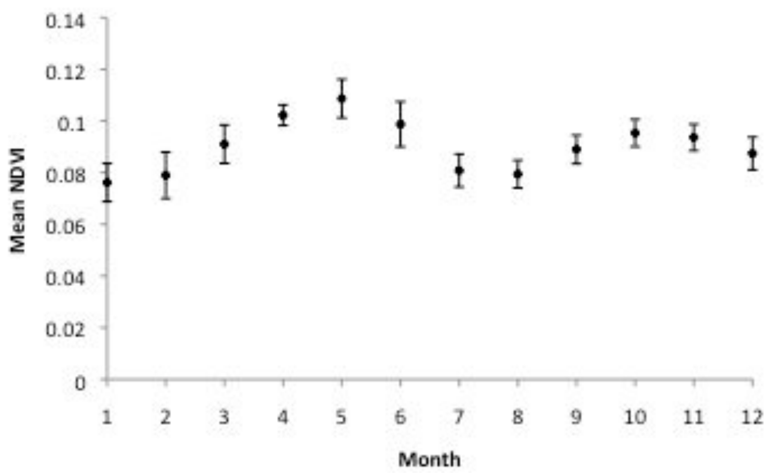
To extend the finer scale analysis beyond the year 2003, NDVI data were used from instruments aboard the Moderate Resolution Imaging Spectorradiometer (MODIS) Aqua and Terra satellites. These data are available from 2002 onwards, at 0.5km resolution. The 'modis' package in R was used to download, mosaic and re-project data for the area of interest, for the period 2001 – 2012.

Additional analyses

Trends in NDVI

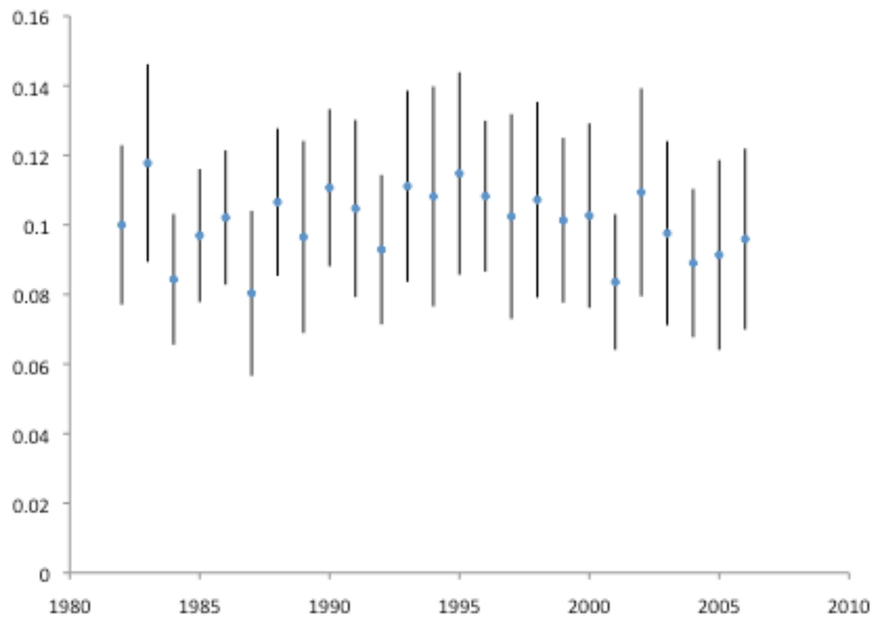
The annual pattern for NDVI in the Uzbek Ustyurt conforms to a pattern that might be expected for vegetation cover in the Northern hemisphere (Fig. A2.2). Again, it is noted that NDVI is not necessarily a useful indicator of vegetation cover in the Ustyurt outside of the spring period.

Figure A2.2*: Mean NDVI values by month for the area of interest, 1982 – 2006. 95% confidence intervals*



Analysis of the GIMMS data set provides an estimate of the trend in mean spring NDVI across the area of interest from 1981 – 2006 (Fig. A2.3). There is no significant increase or decrease in mean annual spring NDVI across this period.

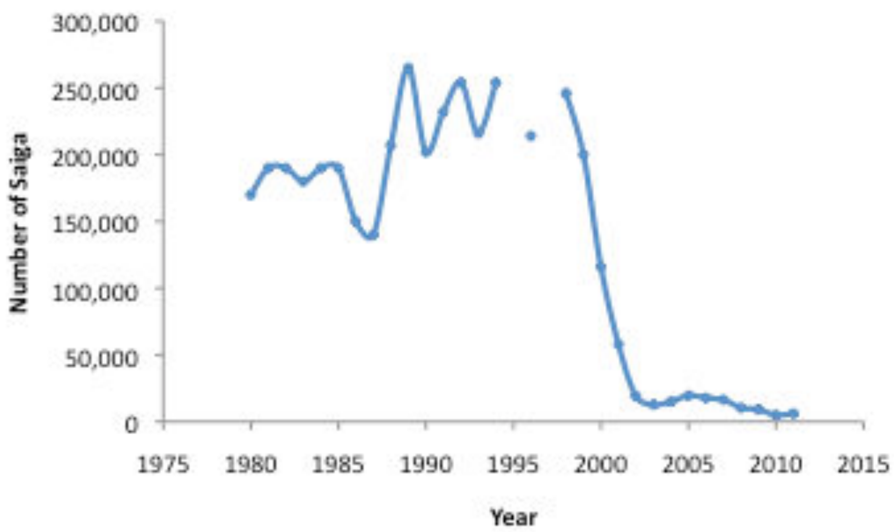
Figure A2.3*: Mean spring NDVI for the Uzbek Ustyurt, 1981 – 2006 from the GIMMS data set. Error bars show the standard deviation*



Protected species

The saiga antelope *Saiga tatarica* has received limited and inconsistent monitoring effort in Uzbekistan. However, the Ustyurt population is contiguous between Uzbekistan and Kazakhstan, and has been monitored more closely in the latter country. Data from the Institute of Zoology (IoZ) Kazakhstan show a drop in recorded saiga numbers of two orders of magnitude in recent decades, consistent with a known collapse in the population (Fig A2.4).

Figure A2.4*: Annual saiga population abundance by year for the Kazakh Ustyurt (IoZ Kazakhstan)*



In the Ustyurt, a number of threatened species exist or are recorded as previously having existed. The Asiatic cheetah *Acinonyx jubatus venaticus*, Caspian tiger *Panthera tigris virgata*, and Transcaspian urial *Ovis orientalis vignei* have all been extirpated from the study area, the Turkmen caracal *Caracal caracal michaelis* has probably been extirpated in some areas, whilst Brandt's hedgehog persists (Ackura et al., 2008). Threatened or endemic mammal species also include the goitered gazelle *Gazella subgutturosa*, central Asian tortoise *Testudo horsfieldii*, desert monitor *Varanus griseus*, and four-lined snake *Elaphe sauromates* (Kashkarov et al., 2008). There is evidence for the occurrence of honey badgers *Mellivora capensis* in the Ustyurt (Asimov et al., 2009). Threatened or endemic bird species include the globally threatened Houbara bustard *Chlamydotis undulata*, cinerous vulture *Aegypius monachus*, eastern imperial eagle *Aquila heliaca*, and great bustard *Otis tarda* (Kashkarov et al., 2008).

Abundance data for saigas were collected during consistently executed aerial surveys overseen by the Institute of Zoology, Kazakhstan, but biases and uncertainties in the monitoring technique mean that the rates of decline may be lower than suggested from the data (McConville et al., 2009). Whilst the Ustyurt saigas spend most of the year in Kazakhstan, from where these data are obtained, a variable proportion of the population migrates to Uzbekistan during the winter months (Bekenov et al., 1998).

It is noted that, whilst the flora and vertebrate fauna of Uzbekistan have been well researched, there are currently few in-country experts on invertebrates. This does not necessarily mean that none exist, but the inaccessibility or such expertise is identified as an important gap.

Climate

Based upon data collected at the Jaslyk meteorological station in the centre of the Uzbek Ustyurt, it is possible to calculate average monthly temperature and rainfall for the period 1977 – 2005 (Fig. A2.5, Fig. A2.6)

Figure A2.5*: Average temperature by month at Jaslyk, 1977 – 2005. 95% confidence intervals.*

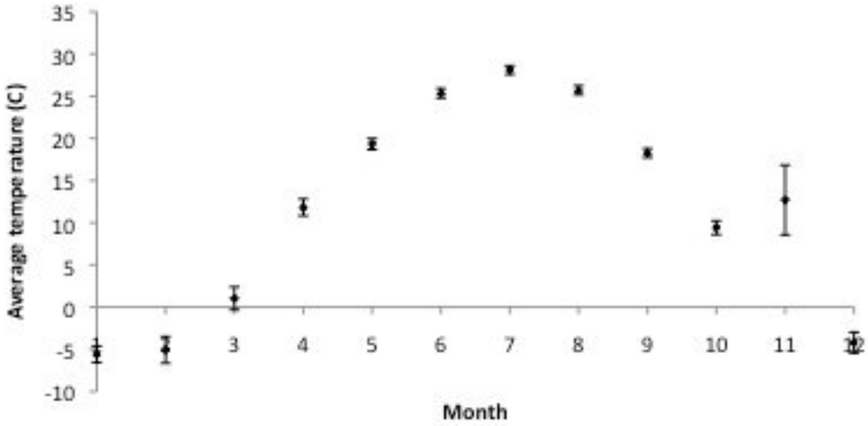
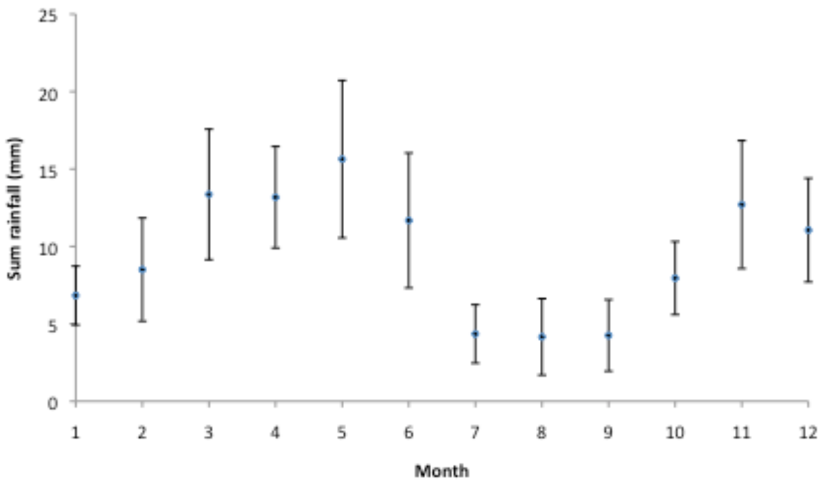
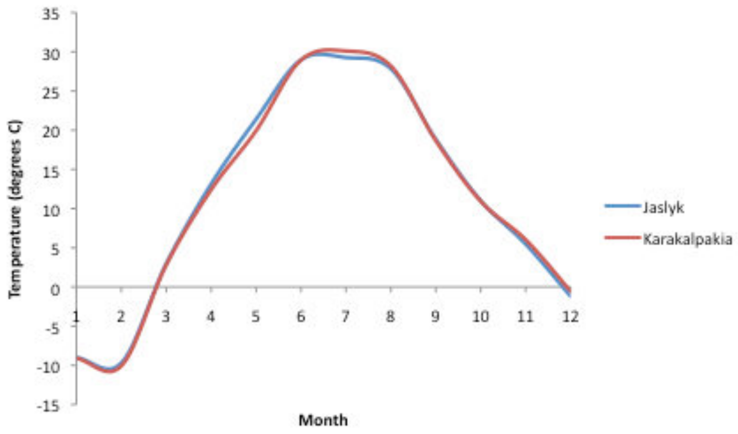


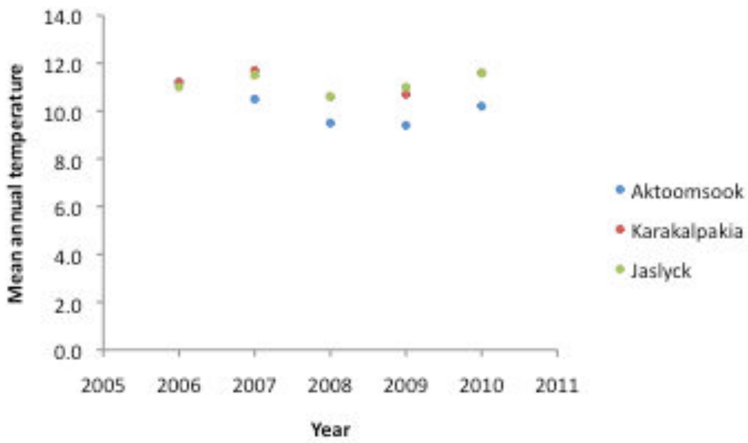
Figure A2.6*: Average sum rainfall by month at Jaslyk, 1977 – 2005. 95% confidence intervals.*



The data in Figures A2.5 and A2.6 were all collected from one location in the Ustyurt, and so do not necessarily indicate climate across the plateau as a whole. However, it is possible to compare temperatures recorded here with temperature recorded at different locations from 2005 – 2010, and these are closely correlated (Fig. A2.7).

***Figure A2.7**: Comparison of temperature data between meteorological stations at Jaslyk (J), Karakalpakia (K), and Aktoomsook (A). (a) 2010 monthly average temperature at J and K. (b) mean annual temperature from 2005 – 2010, A, J and K.*

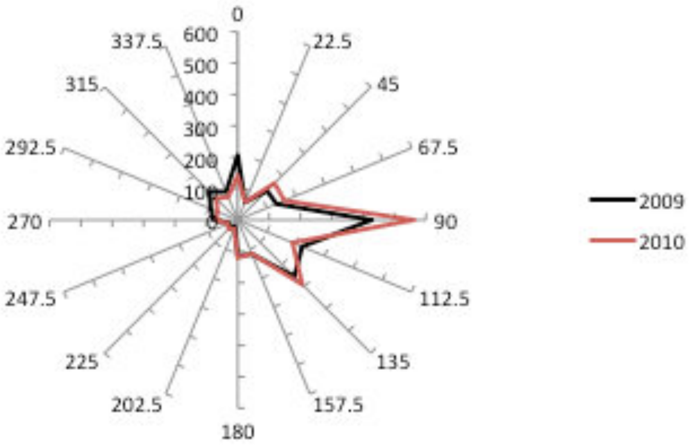




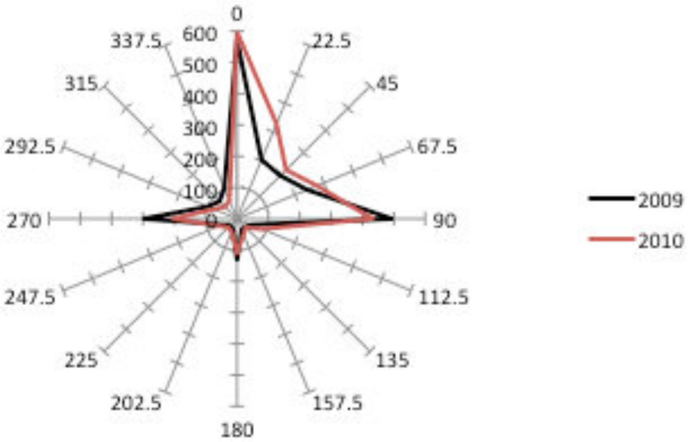
The study area is characterised by strong winds, and dominant wind direction can be an important consideration from the perspective of biodiversity conservation and human health (Micklin, 2007; UNDP, 2010a). Wind direction was plotted based upon data obtained from I. Aslanov, who in turn collected it from the meteorological stations at Jaslyk and the town of Muynak (Fig. A2.8).

Figure A2.8: *Dominant wind direction in (a) Jaslyk and (b) Muynak, based on data obtained from hydro-meteorological stations*

(a)



(b)



Variation in NDVI has been linked to climatic factors in other studies (e.g. Robinson, 2000). It is possible to show, using general linear models (glm), that changes in temperature and rainfall can explain a significant amount of variation in NDVI (Table A2.2).

Table A2.2: *Results of fitting a glm to NDVI with climatic explanatory variables, and a time lag of 0 (t), 1 month (t + 1), and 2 months (t + 2). P-values, with starred significance (0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1)*

	t	**t + 1**	**t + 2**
Mean temperature	8.02e-08 (***)	2.04e-08 (***)	1.99e-05 (***)
Sum rainfall	n/s	2.46e-05 (***)	0.00391 (**)
Interaction (mean temperature and rainfall)	5.29e-06 (***)	0.0483 (*)	n/s

Socioeconomic

The human population of the administrative regions containing the Ustyurt has grown over the last two decades. However, the rate of growth depends upon the region - so that whilst the population of Kungrad continues to expand linearly, that of Muynak has stagnated (Table A2.3). It could be speculated that the lack of population growth in Muynak is linked to its former dependence upon the Aral Sea, as a fishing port.

Table A2.3: *Population of Kungrad and Muynak districts in Karakalpakstan in thousands (Karakalpak administration centre).*

Year	**'000 people in Kungrad district**	**'000 people in Muynak district**
1990	93.3	26.0
1991	95.9	26.5
1992	97.5	27.2
1993	99.4	27.6
1994	101.2	27.7
1995	102.4	27.8
1996	103.8	28.0
1997	106.0	28.1
1998	107.2	28.5
1999	108.6	28.7
2000	110.2	29.1
2001	111.3	29.1
2002	112.1	28.8
2003	112.8	28.7
2004	112.7	28.7
2005	112.6	28.4
2006	112.9	28.4
2007	113.4	28.4
2008	114.7	28.6
2009	115.5	28.3
2010	116.6	28.5
2011	117.0	28.5

Agriculture, primarily pastoral, is a key component of the local economy. Data were obtained from the Karakalpak Ministry of Agriculture upon the number and type of livestock kept in the region in 2010 (Fig. A2.9). Trend data are unavailable, apart from a total figure for the number of cattle kept in Karakalpakstan as a whole – which demonstrated an upward trend according to the official records (Fig. A2.10).

Figure A2.9: *Number of livestock kept in the Muynak and Kungrad districts of Karakalpakstan in 2010*

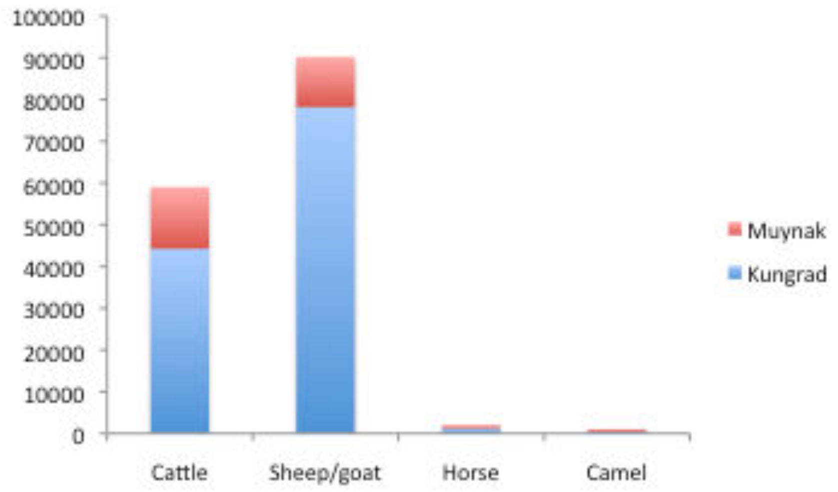
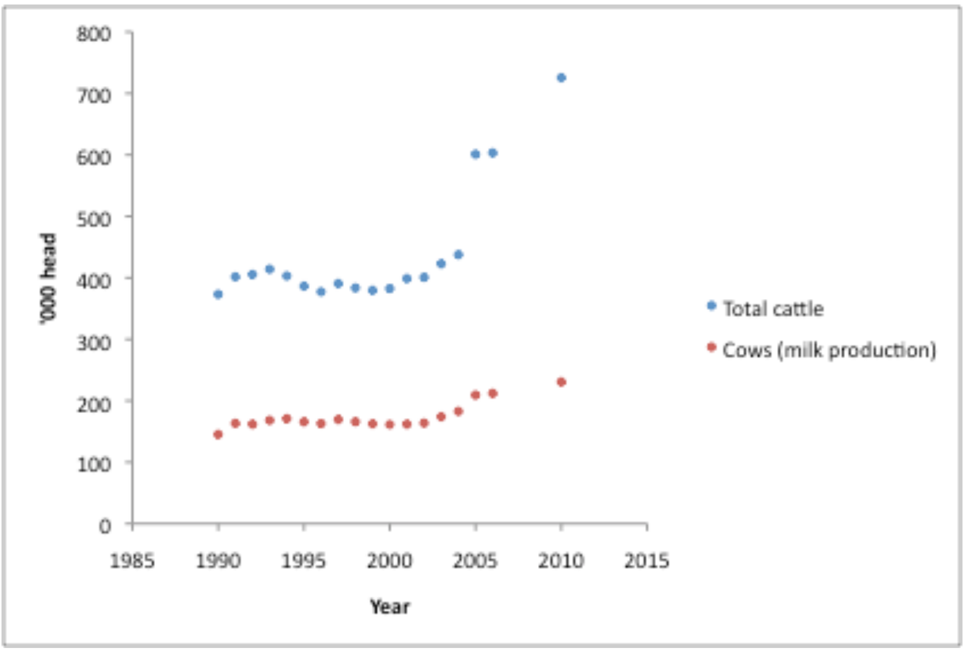


Figure A2.10: *Number of cattle kept in Karakalpakstan in 1988 – 2006.*



The grazing of livestock was observed in the field to have caused extensive clearance of vegetation close to settlements and other infrastructure such as roads and rail (Fig. A2.11).

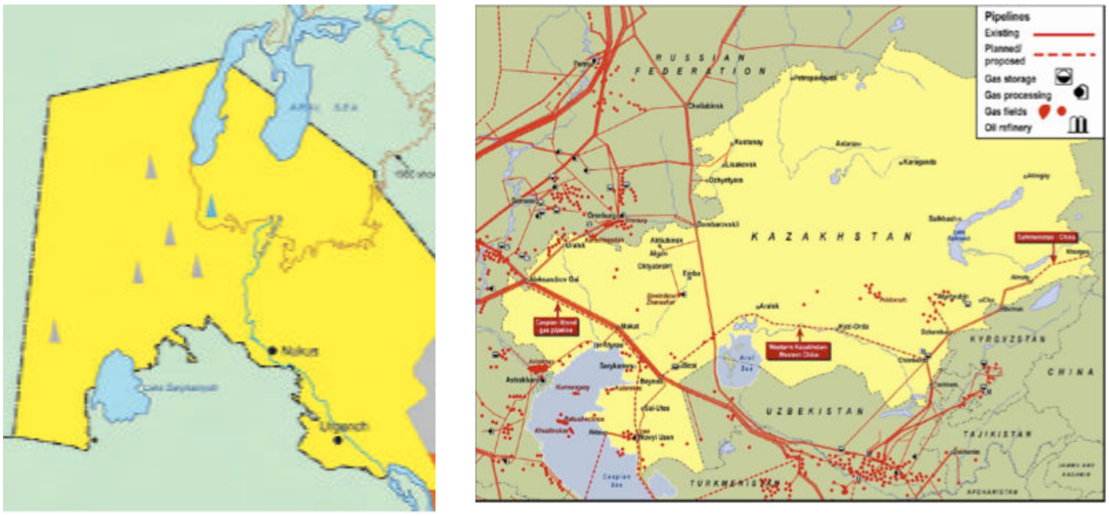
Figure A2.11*: Observed example of vegetation clearance as a result of grazing near a temporary settlement, < 5km from Karakalpakia (photo: J. Bull, 2011).*

Industry

The key commercial sectors in and around the area of interest are extractive (oil and gas, minerals) and transportation (road and the railway). The focus for this thesis is the oil and gas sector, due to the potential use of biodiversity offsets by this industry in Uzbekistan.

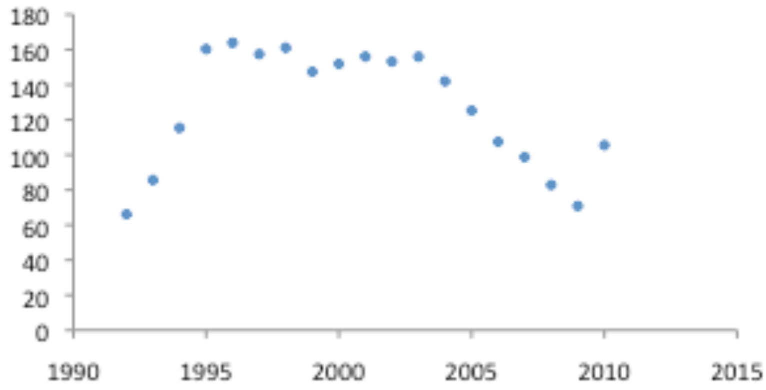
Little detailed information is available in the public domain on the spatial distribution of oil and gas facilities and infrastructure in Ustyurt, but the approximate locations of key components such as extraction sites and main pipelines is known (Fig. A2.12).

Figure A2.12*: (a) Location of gas (blue) and gas condensate (grey) extraction locations in Karakalpakstan (UN, 2010b). (b) location of main pipelines in Central Asia (Yenikeyeff, 2008)*



Data on oil and gas production for the region were also not available, however, official records exist of the annual production of both oil and gas from Uzbekistan as a whole over the last two decades (Fig. A2.13). Natural gas production continues to expand, driven in part by existing and projected exploitation of gas resources within the area of interest.

Figure A2.13: *(a) Oil production, Uzbekistan (thousand barrels per average day); (b) natural gas production, Uzbekistan (billion cubic feet per year) (EIA, 2012)*



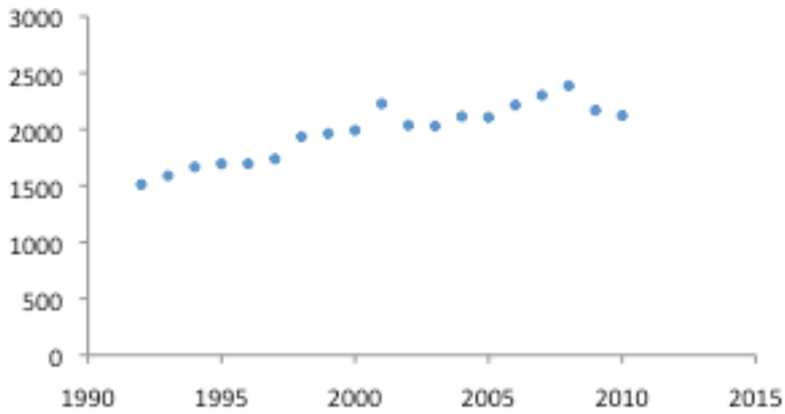
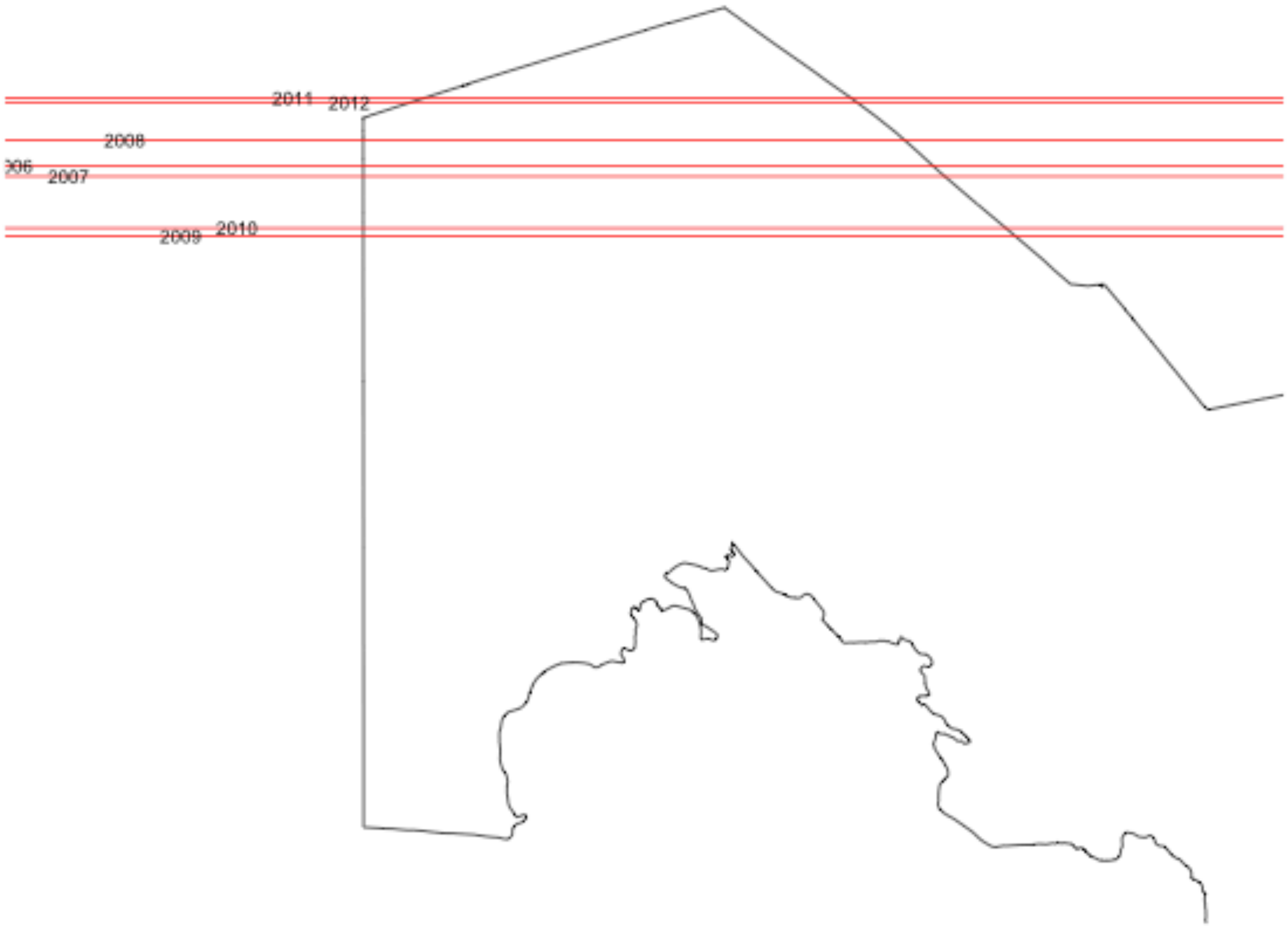


Figure A2.14*: According to all available participatory monitoring data, the mean annual latitude of saiga locations noted in Uzbek Ustyurt for every year from 2006 to 2012.*



Appendix 2: additional references

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Appendix 3

***Table A3.1**: Site components (Habitat Hectares)*

	Component	Maximum score
Site condition	Large trees	10
	Tree canopy cover	5
	Understory strata	25
	Lack of weeds	15
	Recruitment	10
	Organic litter	5
	Logs	5
Landscape context	Patch size	10
	Neighbourhood	10
	Distance to core area	5

***Table A3.2**: Understory strata scores (Habitat Hectares)*

First decision	Second decision	Value
All strata (lifeforms) absent		0
< 50% strata present		5
50 – 90% strata present	Of these, > 50% modified	10
	Of these, < 50% modified	15
> 90% strata present	Of these, > 50% modified	15
	Of these, < 50% modified	20
	Of these, none modified	25

***Table A3.3**: Recruitment scores (Habitat Hectares)*

First decision	Second decision	Third decision	Prop. of benchmark	
			> 50%	< 50%
No evidence recruitment	EVC recruitment not driven by episodic events		0	0
	EVC recruitment driven by episodic events	Evidence of event	0	0
		No evidence of event	5	5
Evidence of recruitment	Proportion of species that have recruitment	< 30%	3	1
		30 – 70%	6	3
		> 70%	10	5

***Table A4.4**: Patch size scores (Habitat Hectares)*

Area	Score
< 2 ha	1
2 – 5 ha	2
5 – 10 ha	4
10 – 20 ha	6
> 20 ha (disturbed)	8
> 20 ha (not disturbed)	10

***Table A5.5**: Neighborhood scores (Habitat Hectares)*

Radius	% native vegetation	Weighting	Score
100m	*input*	0.03	*= % * weighting*
1km	*Input*	0.04	*= % * weighting*
5km	*input*	0.03	*= % * weighting*
		Sub-total	*= sum(above)*
Disturbed			*subtract 2 if disturbed*
		Total	*= total sum(above)*
		Total (rounded)	

***Table A6.6**: Distance to core area scores - subtract 1 if disturbed (Habitat Hectares)*

Distance	Score
> 5km	0
1 – 5 km	2
< 1 km	4
Contiguous	5

***Table A6.7**: Adapted score for lichen cover (modified Habitat Hectares)*

Lichen cover	No high threat	< 50 % high threat	> 50 % high threat
> 50 % of cover	4	2	0
25 – 50 % of cover	7	6	4
5 – 25 % of cover	11	9	7
< 5 % of cover	15	13	11

***Table A6.8**: Distinctiveness scores (UK metric)*

Distinctiveness	Score
High	6
Medium	4
Low	2

***Table A6.9**: Condition weighting (UK metric)*

Habitat condition	Score
Good	3
Moderate	2
Poor	1

***Table A6.10**: Calculate number of biodiversity units (UK metric)*

		Habitat distinctiveness		
		Low	Medium	High
Condition	Good	6	12	18
	Moderate	4	8	12
	Poor	2	4	6

Table A6.11*: Multiplier for delivery risk (UK metric)*

Difficulty restoration/recreation	Multiplier
Very high	10
High	3
Medium	1.5
Low	1

Table A6.12*: Multiplier for spatial risk (UK metric)*

Location	Multiplier
Biodiversity strategy area	1
Buffers, links, restores, expands habitat adjacent to strategy areas	2
No contribution	3

Table A6.13*: Multiplier for temporal risk (UK metric)*

Years to target condition	Multiplier
5	1.2
10	1.4
15	1.7
20	2.0
25	2.4
30	2.8
32	3

Table A6.14: *Values used in calculations based upon Canadian methodology, for % vegetation cover (% cover) and mean species abundance (MSA)*

Variable	% cover	MSA
p_{MOD}	30	0
p_{NOW}	30	1
A_{mod}	0	312.1910082
p_{MAX}	30	1
A_{loss}	220	220
p_{COM}	30	1
p_{NOW}	0	0

Appendix 4

***Table A4.1**: Summary of headline recommendations for the implementation of biodiversity offsets In the Ustyurt plateau, with reference to relevant chapters of this thesis*

Category	Challenge	Suggestions for biodiversity offsets in the Ustyurt	Limitations for biodiversity offsets in the Ustyurt	Relevant chapters
Design	Currency	Use multiple metrics: - Area & condition of vegetation, based on an adapted Habitat Hectares methodology - Suitability of habitat for threatened fauna	- Limited information on disturbance of industrial activity on fauna - No data available on important taxa e.g. invertebrates, microorganisms - No genetic data	6, 7
	No net loss	Given the frame of reference: - Retain current extent and condition of vegetation cover. - Seek increase in saiga antelope population, as an indicator species for wider fauna conservation.	- Limited information available on which to base predicted counterfactual scenarios	4, 5
	Equivalence	- Trade vegetation losses for restored habitat of the same type, nearby (in kind). - Trade losses in suitability of species habitat near industry for gains in suitability of saiga habitat in Saigachy (out of kind)	- No data on costs and effectiveness of vegetation restoration in the Ustyurt - Not clear whether saiga antelope can act as an effective proxy for other species impacts	7, 8
	Longevity	- Consider trends when choosing sites for offsets - Protect restored habitat from destruction or degradation, and saigas from poaching, for as long as the impacts of industry persist - Adapt management to account for long term climate impacts - Use a discount rate of zero	- The environmental, institutional and political status of the Ustyurt has been relatively unstable over, for instance, the last 50 years. Not clear whether this will continue and so threaten persistence of offsets	5, 8
	Time lag	- Expect a time lag of up to 5 years in vegetation restoration (implement offset alongside development) - Expect a time lag of 5 - 10 years for meaningful increase in saiga population (apply a multiplier to fauna species offset requirement)	- A habitat or species bank type solution to time lags is unlikely in the Ustyurt, as there is no private land ownership, and hence the offsets would effectively have to be funded in advance by the State	5, 7
	Uncertainty	- Apply large multipliers to offset requirements to account for uncertainty, using a framework based upon Moilanen et al. (2009) - Seek ongoing improvement in managing uncertainty, based	- Suggested uncertainty framework leads to very large multipliers - Multipliers still widely considered a blunt tool for managing uncertainty	8

		upon any advances in the literature		
	Reversibility	- Losses in vegetation cover can be considered reversible - Any direct mortality of fauna caused by industry, especially saigas, could be considered irreversible and therefore not included in an offset mechanism	- Certain species of plant are on the Red List, and loss of these might be considered irreversible - Not clear to what extent fauna are impacted, either directly or indirectly, by industry	6
	Thresholds	Use framework developed by Pilgrim et al. (2013) to set clear thresholds: - E.g. cumulative loss of vegetation cover > 5 % of the plateau by area is unacceptable, and can no longer be offset - E.g. any direct mortality of fauna species is unacceptable, and can not be offset	- Insufficient data to objectively define ecological thresholds - Thresholds for other taxa (e.g. invertebrates) impossible to set at this stage	5, 6
Implementation	Compliance	- Incorporate biodiversity into environmental impact assessment procedure - Set clear and prescriptive method and guidance for implementing biodiversity offset scheme (i.e. not open to interpretation) - Partially use finances from offset scheme to fund monitoring	- Rule breaking behaviour (e.g. poaching, illegal natural resource use) has historically been a challenge to compliance in the region - Evidence already exists of a lack of rigor in designing offset proposals for the region (e.g. Mott Macdonald, 2012)	5
	Monitoring outcomes	- Partially use finances from offset scheme to fund monitoring - Allow independent organizations (e.g. conservation NGOs) to monitor progress	- Numerous unrelated pieces of country legislation hinder monitoring (e.g. limit on travelling near the Kazakh border) - Lack of international NGOs in the country	5,8
	Uncertainty	- Report results of any existing habitat restoration projects in both public domain and scientific literature - Under training needs analysis for restoration expertise - Collaborate more closely with other saiga range states to understand methods for successfully reducing poaching	- It is unclear whether any expertise exists in habitat restoration in the country - Uncertainties in saiga population abundance and distribution, and response to anti-poaching measures	