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Comparing biodiversity offset calculation methods with a case study in Uzbekistan

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ABSTRACT

Biodiversity offsets are interventions that compensate for ecological losses caused by economic development, seeking 'no net loss' (NNL) of biodiversity overall. Calculating the ecological gains required to achieve NNL is non-trivial, with various methodologies available. To date, there has been no comparison among methodologies for a common case study. We use data on industrial impacts in Uzbekistan to provide such a comparison.

We quantify losses from 40 years of gas extraction, using empirical data on vegetation impacts alongside estimates of disruption to mammals. In doing so, we implement a novel technique by estimating spatial 'functional forms' of disturbance to calculate biodiversity impacts. We then use a range of offset methodologies to calculate the gains required to achieve NNL. This allows a crude comparison of the potential biodiversity outcomes of "in kind" offsets (here, vegetation restoration) with "out of kind" offsets (protecting fauna from poaching).

We demonstrate that different methods for calculating the required offset activities result in divergent outcomes for biodiversity (expressed in habitat condition x area, or 'weighted area'), and different trajectories in biodiversity outcomes over time. An Australian method is currently being considered for adoption in Uzbekistan, but we show that it would require careful adjustments to achieve NNL there.

These findings highlight that the method used to quantify losses and gains strongly influences the biodiversity outcomes of offsetting, implying that offsets generated using different methodologies are not transferable between jurisdictions. Further, conservation gains from out of kind offsets may outweigh those from strict in kind NNL interpretations.

1. Introduction

Biodiversity offsets ('offsets') are a mechanism by which industry can compensate for unavoidable ecological losses associated with development (Madsen et al., 2011). Offsets are implemented through both regulatory and voluntary schemes (Doswald et al., 2012). The essential objective of most offset policies is 'no net loss' (NNL) of biodiversity alongside economic development (BBOP, 2012; Bull et al., 2013a); accepting local losses at the sites of activity but compensating for these by producing equivalent biodiversity gains elsewhere. Offsets should generally be implemented as part of a mitigation hierarchy, such that negative impacts are first avoided and minimised where possible (Gardner et al., 2013). A challenge to effective offsetting, having quantified the residual biodiversity losses associated with development, is the calculation of the biodiversity gains required to deliver NNL (Quétier and Lavorel, 2011; Bull et al., 2013a). Losses and gains are separated in space and time, and potentially differ in biodiversity type; hence there is a need for a common measure of ecological equivalence to compare them.

The term "offset" encompasses a range of approaches to comprehensive (NNL) biodiversity compensation, from habitat-specific calculations to generalisable frameworks (Madsen et al., 2011; Doswald et al., 2012). Several different methodologies exist for calculating the gains required to compensate any given development project: some use area as a proxy for habitat losses and gains (e.g. King and Price, 2004, suggest that many US Wetland Banking offsets effectively use an area-based approach); some use a combination of area and 'functionality' of the habitat (e.g. Canadian Fish Habitat);

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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 (e.g. US Conservation Banking; McKenney and Kiesecker, 2010; Quétier and Lavorel, 2011). Recent developments include a pilot UK policy (Defra, 2011), and a South African policy which incorporates consideration of ecosystem services (Brownlie and Botha, 2009). Some methodologies were developed for specific circumstances, such as those governing native grassland clearances in Victoria, Australia; others, such as US Wetland Banking, are intended as general frameworks.

Despite the underlying NNL objective, it is not clear how such methodologies compare to one another in terms of biodiversity outcomes, when applied to a common case study. Here, we fill this gap, whilst providing a basis for exploring the extent to which different offset methodologies interpret and achieve NNL. Such a comparison is important to highlight to what degree different methodologies are designed within specific jurisdictional contexts, and for different conservation priorities: a point perhaps not always recognized by those designing polices, who might rely heavily upon existing methodologies developed elsewhere when designing their own. The work can also provide insight into how far national offset policies concur on the ecological requirements for NNL, contributing to debate upon whether international offset trades are possible, e.g. trading impacts in one country for offsets in another.

1.1. Objective of biodiversity offsets

Whilst offsets ostensibly seek NNL overall, each approach inevitably focuses upon specific sub-components of biodiversity as proxies for total biodiversity (Bull et al., 2013a) – biodiversity being "the sum total of all biotic variation from the level of genes to ecosystems" (Purvis and Hector, 2000). Offsets can rely upon *habitat-based*, *species-based*, or other calculation methods: respectively, whether offsets focus on vegetation assemblages, focus upon particular species (usually fauna), or consider alternatives such as ecosystem services (Quétier and Lavorel, 2011). We group a set of ecological compensation measures – not all true biodiversity offsets, but which require NNL – into those which are habitat-based or species-based. We do not consider ecosystem service offsets here as, which have yet to become established.

Habitat-based approaches generally rely on measures of area and habitat 'condition' to calculate losses and gains (BBOP, 2012). Victorian native grassland compensation in Australia uses 'habitat hectares', 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 criteria including vegetative recruitment and presence of invasive weeds. A variant on this approach is being trialed in the UK (Defra, 2011).

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

A critical consideration is that offset policies do not always restrict biodiversity trades to being 'in kind'. Whilst trading in kind is encouraged (BBOP, 2012), it has been argued that conservation objectives could sometimes be better served by trading 'out of kind' (Habib et al., 2013). Some policies allow e.g. trading of losses in low value conservation areas for gains in high value areas (e.g. Defra, 2011) or even encourage it (e.g. Brownlie and Botha, 2009); or allow for trading losses in the habitat of one species for gains in that of another (e.g. US Conservation Banking). The extent to which out of kind offsets are acceptable, and how to coordinate this at a landscape scale, are currently open questions.

1.2. Testing methodological approaches against a common case study

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 (Bull et al., 2013b). We use the Ustyurt as a comparative case study, exploring the offset requirements that could have been imposed for O&G infrastructure developed over the last 40 years under a range of methodologies. These insights can be used to inform a biodiversity offsetting project led by the United Nations Development Program – although we do not aim here to advise on the most appropriate methodology for the Ustyurt, a decision which would require consideration of other issues beyond the scope of our study. The research is timely because, at a global scale, many countries (including Uzbekistan, but also e.g. the UK) are developing regulatory frameworks for offsetting.

The Ustyurt plateau (44°N, 57°E) is shared between Uzbekistan and Kazakhstan, west of the Aral Sea. Approximately 100,000 km^2 of the plateau is within Uzbekistan. It is semi-arid and dominated by *Artemisia*, *Anabasis* and *Halyoxylon*, and home to fauna including the Critically Endangered saiga antelope *Saiga tatarica*. There are four small settlements on the plateau, a railway and gas pipelines, and increasing natural gas exploration and extraction activity. Habitat clearance and disturbance to threatened fauna are material ecological impacts of the O&G industry (UNDP, 2010).

Vegetation clearance due to O&G activity has been quantified (Jones et al., 2014), allowing the application of habitat-based offset calculation methodologies. For species-based methodologies, the flagship species is the saiga antelope (UNDP, 2010). This nomadic ungulate previously occurred in large numbers throughout the region, and was the only abundant large herbivore in the ecosystem (Bekenov et al., 1998), potentially having a substantial role in structuring vegetation communities. Saigas have declined by >90% in the region since the early 1990s as a result of poaching (Kühl et al., 2009), making them a conservation priority for Uzbekistan. Human presence and infrastructure have behavioral impacts upon saigas, modifying their use of habitat (Singh et al., 2010; Salemgareev, 2013), but there are no data on these impacts for the Uzbek Ustyurt, and no suggestion that poaching is directly attributable to the O&G industry. To estimate potential disturbance to saigas from O&G infrastructure, thereby developing a species-based calculation method, we use estimates from a meta-analysis into the influence of human disturbance upon mammals.

2. Materials and methods

2.1. Implementing the principles of biodiversity offsetting for the case study

Offset projects require the creation of additional biodiversity value, so those hypothetically implemented in our calculations needed to raise the condition of degraded land in the Ustyurt. 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., 2011; Bull et al., 2014). The Ustyurt habitat has deteriorated in recent decades as a result of the Aral Sea crisis (Micklin, 2007). However, to concentrate on the comparative study of different metrics, we simplify by treating the habitat as stable.

In practice, habitat-based offsets might involve managed habitat restoration, such as reseeding areas in which vegetation had been cleared. For species-based offsets, activities might include reducing non-O&G 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 (Bull et al., 2013b). Note that if we intended to recommend an actual offset policy for the case study region, the spatial distribution of offset projects across the landscape would require consideration, but that is beyond the scope of this study.

Offset requirements were calculated on the simplified basis that all projects take land somewhere in the plateau that has zero condition, but no existing infrastructure, and restore it to pristine levels of vegetation cover or suitability for saigas (i.e. condition = 1). The basis for this from a habitat point of view is that Ustyurt 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). From a species point of view, the suitability of a patch for saigas might be determined by how much hunting takes place there, such that there could be areas where all mammals are hunted (condition approaches 0), but that with a concerted effort to reduce hunting becomes essentially safe (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 is <1% of the region by area, this is not unreasonable.

Known O&G infrastructure in the Uzbek Ustyurt consists of 6 major facilities, 3 main pipelines, multiple off-road tracks, and a railway (Jones et al., 2014). We assume that villages would have existed independently of O&G. The development of O&G 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), so we assume that industrial impacts increased linearly over the 40-year period.

Summing estimated biodiversity losses and offset gains, on development and offset sites only, over the assumed 40-year period, gives a net aggregated biodiversity outcome against a project-scale baseline (Bull et al., 2014). We 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). So, the loss of an area of vegetation could be compensated for by reseeding an equivalent area of condition zero vegetation elsewhere, where condition zero means bare soil (in kind), or by paying for anti-poaching patrols elsewhere, where condition zero means fauna are heavily poached (out of kind). We consider both options (Fig. 1). More generally, offsets can be considered out of kind if far from development sites, but as we do not analyse the spatial distribution of offset sites, this is ignored. In the absence of empirical data and based upon discussion with national experts, a 5-year maturation period was assumed for vegetation restoration activities, whilst anti-poaching measures were assumed to act within a year.

Some methodologies include correction factors for uncertainty. We included two types of uncertainty in our calculations: uncertainty in the area impacted by development (Jones et al., 2014), and the possibility of up to 50% non-implementation of offsets on the ground. The latter is an arbitrary but potentially realistic rate, and not unreasonable given that offset success rates in the literature range from 0% to 74% (Bull et al., 2013a). Uncertainty arises from numerous other sources that we do not consider here (Kujala et al., 2012). An exhaustive framework for quantifying uncertainty in offset projects has yet to be developed (Bull et al., 2013a), although tools for managing uncertainty are under development (e.g. Pouzols et al., 2012). The aim of our exploration was to focus on a comparative study and not upon uncertainty, but we include some uncertainty to demonstrate how it can significantly influence comparative outcomes.

The plateau was considered to be a mosaic of habitat degraded by infrastructure, patches of pristine habitat, and patches of habitat degraded by other influences. We assumed that any patch can potentially be raised to a condition value approaching 1 for both habitat as well as species. Whilst technology that enables the re-vegetation of semi-arid environments has existed for some time (e.g. Gintzburger and Skinner, 1985), the ability to successfully restore vegetation communities as part of offset schemes cannot be assumed, and is potentially a key challenge for offsetting in general (Maron et al., 2012). However, given that our focus here is not whether effective offsetting is possible, we make the assumption that vegetation restoration can be achieved. Similarly, saigas have relatively broad habitat requirements and in the past have been found throughout the potential offset area, so we assume that the opportunity exists for effective saiga protection offsets. We calculated ecological losses based on the impact of known existing industrial infrastructure (Fig. 1). These were then converted into offset requirements (i.e. necessary gains) using each of the regulatory offset policies included in the study (Table 1). An in kind (habitat) offset was considered one in which vegetation losses were compensated for by vegetation restoration, and an out of kind (species) offset was considered one in which vegetation losses were compensated for by gains in suitability of saiga antelope habitat. We assume that the mitigation hierarchy would be similarly applied for all methodologies.

2.2. Calculating the losses caused by oil and gas activities

We 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 than simply using buffer zones, which is common (Osti et al., 2011). Functional forms were generated for both vegetation impacts and mammal disturbance, to allow estimation of losses in these biodiversity components caused by development.

2.2.1. Habitat

Jones et al. (2014) show that vegetation species richness and percentage vegetation cover approach zero within the area occupied by infrastructure on the Ustyurt (i.e. roads, pipelines, extraction sites). However, vegetation is not significantly affected outside of this footprint, with no significant edge effects from e.g. dust deposition. Based upon these empirical data, a step function was 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.

Jones et al. also estimate the total area cleared by O&G on the plateau over the last 40 years = 220 ± 19 km^2. This figure does not account for any potential fragmentation effects, which should be considered in offset schemes, but for which data are not currently available. We use the estimate from Jones et al. as a basis for calculating offset requirements across the landscape. Note that these losses are historical and therefore not likely to be compensated for under any new offset policy. We do not suggest here that an offset policy should account for historical losses (although this has been proposed; Habib et al., 2013), but instead seek to use historical losses in the Ustyurt to compare and contrast existing offset policies.

2.2.2. Species

Disturbance to mammal species from infrastructure presence was used, in our theoretical construct, as a basis for simple out

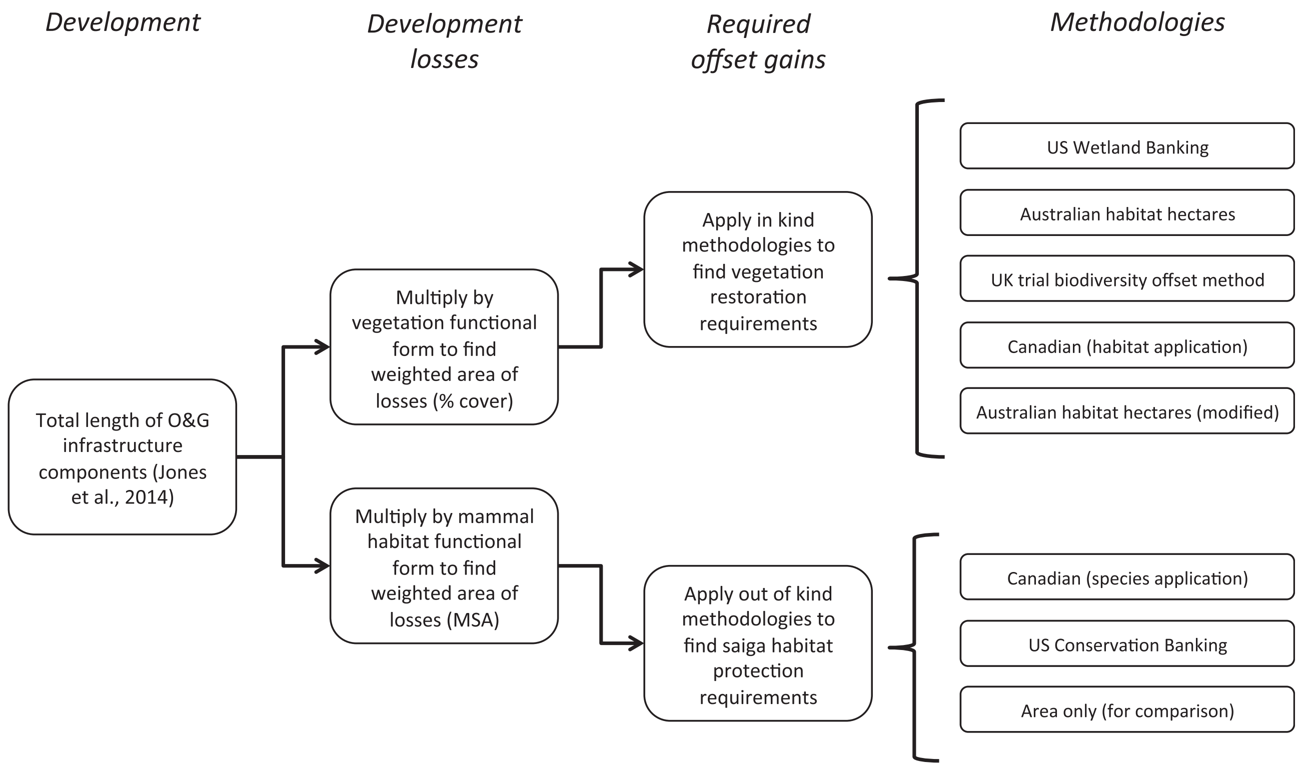

Fig. 1. Flow diagram capturing the basic logic of the approach applied. Taking an estimate of total infrastructure, the functional forms of impact for both in kind (flora) and out of kind (fauna) conservation targets were applied, so as to estimate development 'losses'. These were processed using each different offset methodology to calculate required 'gains'.

Table 1
Regulatory biodiversity compensation policies, with an NNL objective, evaluated in this study.

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)	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)	Compound calculation method; interchangeable 'units' of biodiversity, calculated based on the 'distinctiveness' and 'condition' of the habitat type. Multipliers included	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

of kind offsets. The impact caused by infrastructure upon habitat condition for mammals in general was quantified, and considered equivalent to the amount of compensation required in saiga habitat as part of an out of kind offset. We estimated faunal disturbance, expressed as mean species abundance (MSA) at a given distance from O&G infrastructure, using a meta-analysis of mammal disturbance from infrastructure (Benitéz-Lopez et al., 2010). This meta-analysis was for a range of species and habitats and is not specific to Ustyurt fauna, but in the absence of suitable local data, provided the best available estimate. We 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 focal species for conservation in the region is the saiga (Bull et al., 2013b), the use of a mammalian indicator is logical.

The presence of 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. 2). In a similar habitat and for comparable ungulates (the Gobi, Mongolia), a recent report notes avoidance of infrastructure up to at least ~1 km away (Huijser et al., 2013). This adds weight to our assumption that 5 km is of an approximately appropriate scale of disturbance for saiga-based offsets in the Ustyurt.

The data used to derive a functional form for mammal disturbance (Fig. 2) had a good fit ($R^2 = 0.63$) to the relationship $\text{MSA} = 0.0693 \cdot \ln[x] + 0.2936$, where MSA is 'mammal species abundance' and x is distance (Fig. 3b). The definite integral of this relationship between $x = 0$ and $x = 5000$ (m) gives the graphical area under the curve over that distance (CA_{below}), which is equivalent to condition x actual area (we call this the 'weighted area'). The value of MSA at a hypothetical control site (condition = 1) multiplied by the same distance ($x = 5000$ m) gives the benchmark weighted area $CA_{max} = 5000$. The estimated amount of weighted area lost as a result of the presence of infrastructure is then $(CA_{max} - CA_{below})/CA_{max} = 233.7$. In terms of graphical 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.

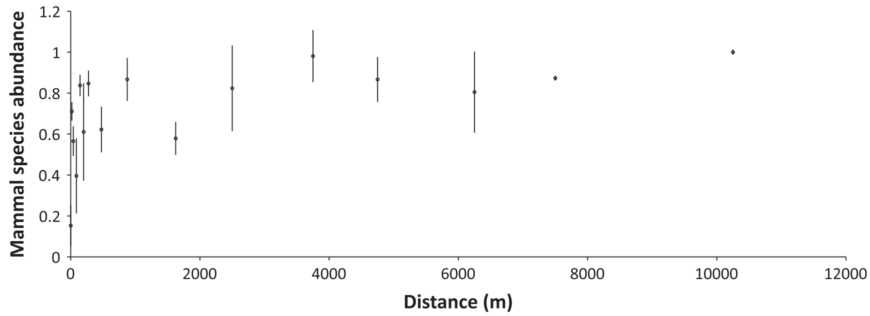

Fig. 2. Plot of mean normalized mammal species abundance (MSA; taking values between 0 and 1) against distance from infrastructure (in metres), using data presented in a meta-analysis by Benitéz-Lopez et al. (2010).

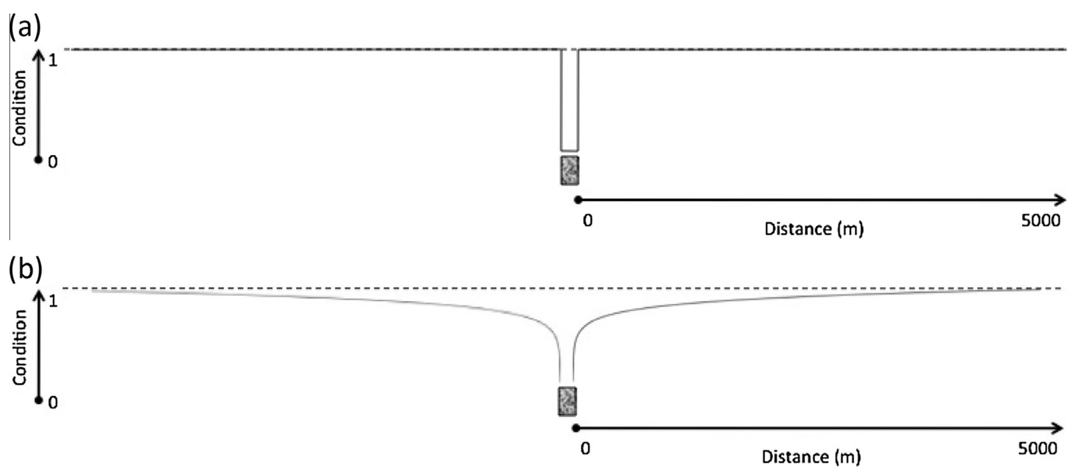

Fig. 3. Functional form of industrial impacts upon 'condition' (a dimensionless index taking values between 0 and 1) for (a) vegetation condition, based upon primary field data (Jones et al., 2014); (b) mammal species abundance (proxy for habitat condition), based upon a meta-analysis (Benitéz-Lopez et al., 2010). Grey block = industrial activity; dashed line = benchmark condition; solid lines = functional form of condition. *Y* axes = condition, *x* axes = distance.

We assume that within the footprint of cleared habitat on the Ustyurt, MSA (and hence condition) = 0, and that it improves outside of this footprint according to the functional form in Fig. 3. We then split the footprint into its separate 'linear' and 'hub' components *sensu* Jones et al. (2014), and calculated the additional area that would be disturbed under the MSA measure for each component, using simple geometry. This results in an overall estimate of 532 km^2 of mammal habitat that had condition reduced from 1 to 0, using MSA as a proxy.

2.3. Offset calculations using different methodologies

Our set of established biodiversity offset methodologies were taken from a recent review (McKenney and Kiesecker, 2010; Table 1). The Brazilian Forest Code was excluded as its future is unclear (Madsen et al., 2011), and it arguably does not fulfill criteria for an offset policy (e.g. Bull et al., 2013a). For the sake of comparison with more established methodologies, we included the approach being developed in the UK (Defra, 2011). We also developed a locally adapted version of the Victorian approach, which is under consideration for use in the Ustyurt by Uzbek authorities (Parkes et al., 2003). Below, we outline the steps for each method, and then describe how we apply them in the Ustyurt. Note that the methodologies applied here are highly simplified versions of those implemented in practice, and we have attempted to capture core underlying principles of each method only. It is the core principles we wish to compare, and further, the complexity of including the numerous variations upon these methodologies used in practice would require more detail than could be included in one manuscript.

In all cases we express the gains required for NNL under a given methodology in condition *x* area (i.e. weighted area), so that we are able to compare biodiversity outcomes. The weighted area is a quantity obtained by multiplying the condition score (which is calculated using a method specific to each given methodology) by the area in km^2 that needs to be offset (which depends only on whether the offset is in kind or out of kind). We make the different gains calculated under each methodology comparable, for our purposes, by simply assuming that areas currently at 0 condition are restored to the benchmark condition of 1. Doing so gives comparative results in units of km^2. A more realistic approach would be to choose areas with intermediate pre-existing condition scores to offset under a range of criteria in a systematic conservation planning approach.

In addition to a static calculation of aggregated landscape-level offset requirements, we incorporate a temporal element by plotting development losses and offset gains through time. To do so, we assume a constant rate of loss from development over a 40-year period, with offsets implemented the year after associated losses were caused. Again, we assume vegetation restoration takes 5 years to mature, and that species offsets take effect immediately.

2.3.1. US wetland banking

Offset requirements are in many cases calculated based upon area of wetland lost, or length of linear features such as narrow waterways (US ACE et al., 1995). Whilst there exist a variety of approaches to wetland mitigation in the US, many of which incorporate biophysical indicators as well as area, here we simplify by treating the approach as purely area-based. This is in line with the observation that the majority of wetland banks use a 1:1 ratio by area for wetland mitigation (King and Price, 2004).

Use in this study: Biodiversity offsets were required to replace the same area of vegetation impacted by development (=220 km^2; Jones et al., 2014).

2.3.2. 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. the 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 we ignored this stage, and treated the plateau as one homogenous habitat type. However, in developing a full offset policy for the Ustyurt, it would be possible to define different vegetation associations.
(2) *Assess condition of different ecological categories against pristine EVC condition, giving each a score up to the maximum* (Table S1.1). Categories in Table S1.1 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. The category for 'organic litter' was also ignored, as this is not particularly a feature of the region, as was 'lack of weeds', as non-native weeds are not a key threat in this habitat. Details of scoring for the other categories are shown in Tables S1.2–S1.6.
(3) The assessment in step 2 provides the 'habitat score'. This is multiplied by the area of the habitat patch to obtain score 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. In reality, various multipliers are then also applied to offset requirements, but here we focus on the basic HH metric itself.

Use in this study: to calculate losses, the area of land impacted by industrial activities (220 km^2; Jones et al., 2014) was assumed to have started at benchmark condition (=1). We completed all calculations in km^2 rather than hectares, making this equivalent to 220 "habitat km^2" impacted by development. The condition score of impacted land was calculated using Tables S1.1–S1.6 and multiplied by 220 km^2, giving a weighted area loss from development of 124.6 units. The offset requirement, as per our 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, i.e. 124.6 km^2.

2.3.3. Adapted Victorian native grassland (Australia)

We created an adapted version of the HH methodology, specific to the Uzbek case study. All categories from application of the existing methodology were retained. The 'lack of weeds' category was replaced with cover of black lichen (*T. desertorum*), which is a sign of undergrazing and prevents vegetative recruitment (G. Gintzburger, pers. comm.) (Table S1.7).

Use in this study: the adapted methodology was applied exactly as for the standard version (Parkes et al., 2003), with the exception that the new category for lichen cover was included in the calculations. The offset requirement calculated under this modified method is 227 km^2.

2.3.4. UK biodiversity offset pilot

Taken from Defra guidance (2011), this offset metric is currently employed in a UK pilot. It is based upon condition and area, incorporates 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 S1.8).
(3) *Apply weighting for habitat condition* (Table S1.9); note that the approach to weighting habitat condition is currently not defined.
(4) *Combine distinctiveness and condition* (by multiplication) to give number of biodiversity units per hectare (Table S1.10). 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 S1.11). Risk is evaluated using figures in a technical appendix (Defra, 2011). No habitat types in the UK correspond directly with those in the Ustyurt, although see Gintzburger and Skinner (1985), where seedling viability increases from 9% to 60% with restoration of semi-arid desert. We 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 S1.12).
(7) *Establish temporal multiplier to account for e.g. delays in restoration*. Again, defined in the technical appendix (Defra, 2011), but no UK habitat types correspond with the Ustyurt. We assume about 3–5 years until maturity with restoration, and >40 years without.
(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: We assigned the pre-development Ustyurt habitat a Distinctiveness score = 6 (high) and Condition score = 2 (moderate). This gives benchmark habitat a score of 12 units per hectare under this methodology, or 1200 units per km^2. The area of habitat affected by development = 220 km^2. Given that the vegetation is effectively cleared over this area, this suggests 1200 * 220 = 26,400 habitat units are lost through development. The multipliers we apply to this total are 1.5 (restoration risk; Table S1.11), 1 (spatial risk, Table S1.12), and 1.2 (temporal risk; Table S1.13), meaning that 475,200 habitat units are required. 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 1200 units per km^2, which would necessitate 396 km^2 of offset project.

2.3.5. Canadian fish habitat

The objective for Canadian fish habitat offsets is NNL in "productive capacity" (DFO, 2002). Although not mandatory, 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:

$$\Delta P_{NOW} = [p_{MOD} - p_{NOW}] \cdot A_{MOD} - p_{MAX} \cdot A_{LOSS} + [p_{COM} - p_{NOW}] \cdot A_{COM} \quad (1)$$

where ΔP_{NOW} is the net change of natural productivity of fish habitat, A_{LOSS} is area of habitat lost due to development activity, A_{MOD} is area modified, directly and indirectly, as a result of the development activity, A_{COM} is area created or modified elsewhere to compensate for the development activity, p_{MAX} is maximum potential unit area productivity rate (or productive capacity), p_{NOW} is present unit area productivity rate, p_{MOD} is modified unit area productivity rate in affected areas, p_{COM} is the compensation unit area productivity rate in affected areas.

ΔP_{NOW} is required to be $\geqslant 0$ for NNL. p_{MAX} is set to 1, and all others are proportions of this.

Use in this study: To achieve NNL, we rearranged Eq. (1), setting $\Delta P_{NOW} = 0$. Consequently, we required that: {area * productivity of lost habitat} + {area * change in productivity of modified habitat} = {area * change in productivity of offset habitat}.

The method could feasibly be applied both to habitat condition for mammal species, and to the condition of vegetation. As a result, in this exploration, we use the Canadian methodology as both a habitat-based and a species-based metric. As measures of productivity, we used the proxies of '% vegetation cover' and 'mean species abundance', for vegetation and fauna respectively, with functional forms for disturbance given in Fig. 3. Parameter values and justifications are listed in Table S1.14. These calculations result in an offset requirement of 220 km^2 under the habitat metric, and 532 km^2 under the species metric.

2.3.6. US Conservation Banking

There is no standard methodology for calculating credits (i.e. offset requirements) for the species-based approach of Conservation Banking; "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...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: We take this criterion to mean that development impacts result in equivalent area of offset habitat required. Here, our exploration bases the analysis upon mean species abundance (MSA) of mammals as described in Section 2.2.2. 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.

3. Results

3.1. Comparing different offset metrics

Aggregated offset requirements calculated using different methodologies varied substantially (Table 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 total amount of mammal habitat that is influenced by the development and ignoring the functional form of disturbance (=9023 km^2), the variation is much larger ($n = 7$, $\mu = 1578$ km^2, $\sigma = 3286$ km^2, $CV^* = 5.73$). However, all results generated using functional forms are reported in weighted area, which although in units of km^2 (i.e. condition multiplied by area in km^2), must be compared to area only measurements with caution. Further, good practice is that area alone should not be used as a metric in offset schemes (BBOP, 2012).

The various habitat-based methodologies result in a range of positive and negative net biodiversity outcomes evolving over the 40-year period, with none exactly compensating for biodiversity losses throughout (Fig. 4). Most result in negative biodiversity outcomes, 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 negative outcome, but after 12–13 years of development attains a neutral outcome for biodiversity, and then goes on to overcompensate for losses, assuming full compliance. The time lag in compensating for losses through the UK method is a result of the assumed delay in restored vegetation maturing, but the inclusion of multipliers eventually results in gains being greater than losses. The Victorian calculation method delivers insufficient compensation and so results in increasing net loss. The adapted Victorian calculation method came closest to achieving a neutral outcome for biodiversity – ostensibly the aim under a NNL objective – although it had still not been achieved after a 40-year period using our assumptions.

The divergent outcomes obtained using different methodologies are likely to be due to: (a) differences in interpretation of ecological equivalence; (b) the calculation methods being overly specific to certain habitats and therefore not adequately capturing Uzbek features (e.g. the non-modified Victorian method); (c) multipliers being explicitly built into calculation methods; and (d) the assumed time lag in habitat restoration gains.

3.2. In kind vs. out of kind offsetting

Results obtained using in kind habitat calculation methods generally differ from those using out of kind species based methods, as expected. The weighted area over which offset activities should take place is larger for out of kind offsets (Table 2). The Canadian and US Conservation Banking calculation methods suggest equivalent requirements for species offsets, both resulting in a net gain in the area of benchmark condition land (Table 2, Fig. 4). This net gain is partly a facet of the equivalency scale we use, derived from the fact that fauna are potentially impacted over a wider area than flora in this specific case study.

4. Discussion

4.1. Comparing different offset metrics

A key finding is the substantial divergence in biodiversity outcomes when using different metrics for calculating the gains required to offset the same development. The specificity of the Victorian methodology to Australian grassland explains the low restoration estimate for the standard version of this method. In

Table 2
Comparison of aggregated offset requirements across different methodologies, expressed as the weighted area within which conservation actions must be applied, under a static appraisal of 40 years of O&G development. Uncertainty represents the potential range in spatial extent of O&G infrastructure across the Ustyurt. "Saiga habitat" is, for context, the area of a proposed reserve available for restoration of habitat condition under species-based offsetting.

Offset policy	Target	Weighted area (km^2)	Uncertainty (km^2)
1. Area only (US)	Habitat	220	±19
2. Area and condition (Victoria)	Habitat	125	±11
3. Area and condition (UK)	Habitat	396	±34
4a. Area and functionality (Canada)	Habitat	220	±19
4b. Area and functionality (Canada)	Species	532	±46
5. Area and condition (US)	Species	532	±46
6. Area only	Species	9023	±779
7. Area and condition (Victoria adapted)	Habitat	227	±20
Proposed Saigachy reserve	Saiga habitat	7352	n/a

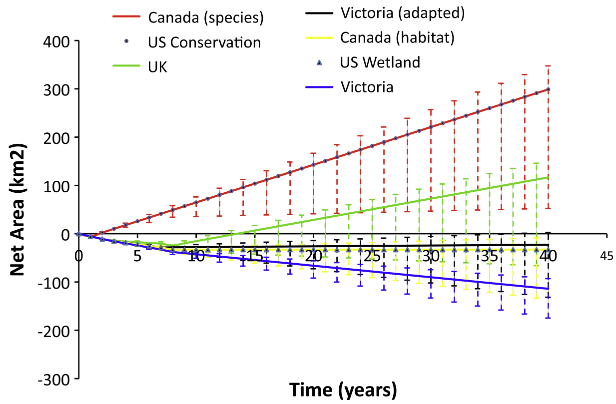

Fig. 4. Plot of net weighted area of land at benchmark condition (in km^2) against time (in years) resulting from hypothetical offsets in the Ustyurt, using different methodologies (Table 1). The Canadian method applied to species (4b in Table 1) and US Conservation (5 in Table 1) methods are exactly aligned, and represent 'out of kind' offsetting here. Upper and lower bounds reflect uncertainty in both estimation of impacts (cf Table 2) and, for the lower bound, the possibility of up to 50% non-compliance.

particular, the Ustyurt scores highly under this calculation method for a lack of invasive weeds: an important problem for Australian grassland but less so in the Ustyurt. Whilst this is perhaps not surprising, given the method was designed specifically for Australia, it is nevertheless pertinent because Uzbek policymakers are currently considering habitat hectares for use in the Ustyurt (J. Bull, pers. obs.). The UK calculation method required a large offset area due to the enforced use of multipliers. 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.

The modified Victorian metric came closest to achieving neutral biodiversity outcomes under the assumptions that we 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 we have modelled here, the difference in 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 our limited treatment of uncertainty results in a large overlap between otherwise divergent outcomes (Fig. 4). An interesting further study would be to repeat our analysis using a tool such as RobOff, which allows a detailed consideration of uncertainty. Doing so would enable the study to be extended by e.g. examining the importance of uncertainty in the response of conservation targets to management, or considering the relative costs of different management actions (Pouzols et al., 2012).

The divergence in outcomes is informative regarding any debate on trade in biodiversity credits between areas using different methodologies for calculating offsets. This could include different countries, but also areas with different ecosystems or types of industrial impact. 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. This echoes recent debates in the literature, which have argued that different jurisdictions might have different thresholds for what would be considered offsetable (Pilgrim et al., 2013; Regnery et al., 2013). Whilst trans-jurisdictional credit trades have yet to be officially proposed, this is a topic of debate amongst offset researchers and practitioners.

4.2. Out of kind offsetting

Restoration of vegetation in the Ustyurt is not necessarily the most urgent conservation priority in the region. Not only does the area impacted by industry constitute $\ll$1% of the plateau (Jones et al., 2014), the vegetation in the region is undergoing wider decline in any case due to unrelated factors (Micklin, 2007). As such, vegetation restoration efforts on the scale potentially required by offset schemes, as calculated here, would arguably do little for regional biodiversity 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 developed an extremely basic scale for equivalence between losses in vegetation and gains in undisturbed mammal habitat. A more nuanced scale might enable out of kind trading between 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 Ustyurt fauna could result in a net gain from a conservation point of view, as the latter is a far more urgent conservation priority – particularly saigas. An exploration of relative costs would be required to ascertain the most cost-effective approach. Furthermore, there are other Ustyurt species, such as threatened birds or reptiles, which could also form the focus of any regional out of kind offsetting scheme. Using different species would probably change the outcomes obtained using the various metrics, not least due to the differences in functional form of disturbance (cf Benitéz-Lopez et al., 2010). However, we do not explore this further here. 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), and it is not unfeasible that this would be similar in 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 (Brownlie and Botha, 2009; McKenney and Kiesecker, 2010), but risks blurring the line between '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, 2010; Bull et al., 2013b), is poaching by those not involved in the O&G industry (Kühl et al., 2009). 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. We stop short of recommending that out of kind offsets are appropriate for the Ustyurt, because there are numerous other topics to consider (e.g. such as impacts upon common species not included in the offset scheme, and stakeholder views), but do consider this a potentially fruitful avenue for further study.

5. Conclusions

We used data on industrial impacts for an Uzbek case study, to compare the range of biodiversity offset requirements generated using different offset methodologies. This included giving consideration to the functional form of disturbance from industrial infrastructure, an approach not previously discussed in the offset literature.

It was found that methodologies from different countries, despite having a common NNL requirement, could result in a range of biodiversity outcomes – highlighting how variable interpretations of NNL can be. This result indicates that trading biodiversity credits between areas using different methodologies would be problematic. Finally, the potential for out of kind biodiversity offsets to result in better conservation outcomes than strictly in kind trades, in certain circumstances, was discussed.

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; it also requires a clear decision as to the fundamental objective of the offset policy.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.biocon.2014.07.006.

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