Time discounting in biodiversity offsets

Summary

Biodiversity offsets are increasingly employed to compensate local development impacts, often through habitat restoration. One proposed correction for time delays in habitat creation involves applying time discounting to future biodiversity "gains". The arguments for this are borrowed from utility discounting in economics. This paper revisits the issue, critically assessing the justifications for discounting presented in the literature (and the choice of rate). I find that discounting in offsets is riddled with conceptual holes, lacks even the most basic justification for the vast majority of values society has for biodiversity, and draws on somewhat bankrupt economic theory that is barely further developed in terms of justification and consistency. While the practice is seductive due to its simplicity and common sense, a better strategy would be to establish a more coherent and firm position against time lags in offset transactions.

Introduction

Biodiversity offsets are increasingly employed under different auspices in different world regions (e.g. endangered species banking, wetland mitigation and tradable development permits; ten Kate et al. 2004). Offsets attempt to compensate local development impacts (at the "impact site") using "gains" in biodiversity at an "offset site", often through habitat restoration. The ratio of biodiversity value (usually expressed in habitat area) gained to value lost is quantified using "offset ratios". Offset ratios may, but usually do not, include additional corrections ("multipliers") for time delays, and one suggested correction involves applying time discounting to future biodiversity "gains" created through habitat restoration (Moilanen et al. 2009, Laitila et al. 2014). A conceptual basis for comparing aggregate, discounted biodiversity value in offset transactions (termed "net present biodiversity value"; NPBV) has also been proposed by Overton et al. (2013). The arguments are borrowed from utility discounting and environmental cost–benefit analysis (CBA) in (neoclassical) economics, but the discount rate is applied to an index of biodiversity value, rather than (revealed or contingent) monetary values. While Overton et al. (2013) attempt to provide broad reasons for discounting biodiversity over time, a theoretical basis for the exact choice of discount rate or function remains elusive (similar to the situation in economics). This paper revisits the issue, critically assessing the justifications for discounting presented in the literature (and the choice of rate) in the context of environmental ethics and ecological economics.

If you can't beat 'em, join 'em!

Time discounting in offsets (and economics) is controversial because it assumes current losses and discounted, but as-yet unachieved, future gains are qualitatively equivalent and comparable (Overton et al. 2013). This is dangerous given the potentially irreversible consequences (e.g. species extinction) of mismanagement (Bekessy et al. 2010) and very poor success record of existing offsets and restoration projects (Kihslinger 2008, Burgin 2010, Bull et al. 2012, Moreno-Mateos et al. 2012, Maron et al. 2012, Curran et al. 2014a). Rather than take an adversarial stance from an ecological perspective, some authors pragmatically suggest applying corrections (offset ratio “multipliers”) so that, at a minimum, robust discounting and uncertainty framework can replace currently arbitrary ratios. This would at least make blatantly unfair trades and net losses of biodiversity transparent to public scrutiny (Overton et al. 2013).

Why discount?

Overton et al. (2013) provide five reasons drawn from service discounting in Habitat Equivalency Analysis (Dunford et al. 2004) and utility discounting in economics (Frederick et al. 2002): (1) the *risk of non-delivery* of future conservation gains, (2) the *lost opportunity cost* of the use of biodiversity (i.e. the debt cannot be “used” by humans until it is recreated) (3) the *rate of return on biodiversity capital* (i.e. the reduced capacity for population reproduction, growth and speciation leading to reduced value to humans in the future), (4) *changes in marginal value* of biodiversity to humans, (5) *pure time preferences* (i.e. individuals favour values now rather than in the future).

These justifications are problematic because they do not clearly differentiate between *anthropocentric* (use and non-use) values and *ecocentric* (intrinsic or moral) values (Palmer et al. 2014), nor define the role of offsets within these value systems. This lack of definition can also be found in the definitions of offsets used by practitioners, such as that of the Business and Biodiversity Offsets Programme (BBOP), which defines the goal of offsets as achieving a no net loss or net gain “with respect to species composition, habitat structure, ecosystem function and people’s use and cultural values associated with biodiversity" (BBOP 2012). Whether to discount, and choosing the appropriate rate, will depend on the type of values considered.

Anthropocentric values for biodiversity also make up a component of the BBOP definition of offsets (“...people’s use and cultural values associated with biodiversity”), and appear in similar definitions for other policies. This component of value probably presents an insurmountable challenge, given the complexities of social value systems. Monetary values might be derived for ecosystem service estimates and a Kaldor-Hicks criterion included, yet other intangible values will always be lost, and the loss will be masked by the apparent validity of the transaction. Chan et al. (2012) recognize eight dimensions of environmental value, ranging from market-mediated exchange values to “transformative” and “metaphysical” (spiritual) values. Because of the context-specific, incommensurable and intangible nature of these values, any attempt to apply offsets should be abandoned. The issues are simply too complex, fundamental uncertainties pervasive and transaction costs likely enormous (beyond anything the trading parties would be willing to pay).

Ecocentric ambiguity

If we restrict discussion to ecocentric values, self-contained or intrinsic value based on ethical extensionism (i.e. that other species may be sentient, feel pain, enjoyment, have a viewpoint etc.; Chan 2011), must largely be ignored as these properties are specific to the individual organism, and not the species as a whole. Thus perhaps only justifications of “relational” intrinsic values would logically apply (i.e. the consideration of value of a biodiversity entity in relation to the interests of other biodiversity entities; Palmer et al. 2014). In this light, we can then appraise the reasons for discounting presented by Overton et al. (2013), of which only three appear to link to relational ecocentric values: non-delivery risk, rate of return on biodiversity capital and the debt-induced opportunity costs. Ideally, discount rates for these components should be based on (objective?) ecological variables reflecting changes in the state of ecological health and diversity over time (i.e.

reflecting how the biodiversity unit affected contributes to the health of biodiversity as a whole).

Risk of non-payment. If the risk of failure grows over time, it could be appropriate to represent it in the discounting term. However, a binary risk (such as for restoration failure) is likely more a function of area and management regime (i.e. planning variables), and thus best dealt using a robustness correction rather than discounting (Moilanen et al. 2009, Overton et al. 2013).

Opportunity cost. The opportunity cost effect could be thought of as the direct debt/delay of relational ecocentric value caused by the offset. While Overton et al. (2013) recommend a range of 1–2%, no justification is provided. One possible alternative would be to base the rate on the proportion of total biodiversity value in the landscape that is impacted by a project. Logically, if there is a background rate of biodiversity loss, then the proportion impacted will grow relative to a decreasing regional “pool” of biodiversity. Thus the rate should grow over time at the rate of estimated habitat loss. The establishment of an appropriate scale at which this background rate is measured, and its measurement is obviously riddled with uncertainty (Brown et al. 2007)., and likely open to favourable interpretation by parties engaged in the trade (Curran et al. 2014b)

Rate of return. The “rate of return” component presumably reflects the rate at which marginal units of biodiversity replenish and grow (including over evolutionary timescales). This might reflect the contribution of the impact site to the maintenance and growth of conservation value in the landscape (e.g. as a source for emigration to other areas). This is often reversed for restored areas, which act as sinks for many threatened species until they reach a late stage of succession, even then exhibiting reduced diversity across a range of indicators (Gibson et al. 2011, Curran et al. 2014a). Overton et al. (2013) again recommend a range of 1–2% without justification. I would propose a that more appropriate estimates would be derived from further research integrating life-history information, meta-population dynamics (e.g. Hanski 2011), evolutionary history (e.g. Jetz et al. 2014) etc. Naturally, only a tiny subset of species will have such information available (most probably only well studied species of larger vertebrates), again leading to massive information gaps and uncertainties. Ambiguities over what exactly is being measured and how rates are derived is likely to provide a huge range for interpretation, again providing ample opportunity for following vested interests, as is currently wide-spread in existing offset policy (Walker et al. 2009, Clare and Krogman 2013).

Conclusions

In economic cost–benefit analysis, discounting of monetary values is often applied based on the rate of return on capital, approximated in interest rates (e.g. on short-term risk-free government bonds; Gowdy et al. 2011). This assumes a principal of weak sustainability, where increased wealth can compensate for environmental costs shifted to the future (Gowdy et al. 2011, De Groot et al. 2013). In the U.S., the National Oceanic and Atmospheric Administration recommends a discount rate for Habitat Equivalency Analysis (HEA) close to the recommended interest rate for environmental valuation (Dunford et al. 2004). This is problematic even from a weak sustainability perspective as estimates of wealth increases do not take into account environmental externalities or social costs (TEEB 2008, Gowdy et al. 2011). HEA also applies these discount rates, not to monetary ecosystem service estimates, but to ecosystem compositional and functional indicators, raising issues of comparability and relevance. In U.K. offset policy, a similar basis for the rate is recommended

(DEFRA 2012). While this makes little sense and has no theoretical basis, I have shown in this article that aiming for more ecologically-inspired rates also hardly seems feasible given the poor state of knowledge on most aspects of biodiversity. Discounting in offsets is thus riddled with conceptual holes, lacks even the most basic justification for the vast majority of values society has for biodiversity, and draws on somewhat bankrupt economic theory that is barely further developed in terms of justification and consistency. While the practice is seductive due to its simplicity and common sense, a better strategy would be to resist the temptation altogether, and establish a more coherent and firm position against temporal imbalances in the loss and gain of biodiversity in offsets (e.g. Bekessy et al. 2010).

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