Ecological Economics 159 (2019) 54–67

Contents lists available at ScienceDirect

Ecological Economics

journal homepage: www.elsevier.com/locate/ecolecon

Analysis

Economic Insights in Ecological Compensations: Market Analysis With an Empirical Application to the Finnish Economy☆

Johanna Kangas*, Markku Ollikainen

Department of Economics and Management, University of Helsinki, Latokartanonkaari 5, P.O. Box 27, 00014 University of Helsinki, Finland

A R T I C L E I N F O

JEL classification:
Q21
Q28
Q57

Keywords:
Biodiversity
Offsetting
Trading ratios
Market equilibrium
Uncertainty

A B S T R A C T

Biodiversity is decreasing at an alarming rate. Current policies focus on the most valuable species and habitats but cannot stop the degradation occurring in less valuable habitats. One solution is offsetting the loss of biodiversity caused by development projects. The basic idea of biodiversity offsetting is simple: a developer must provide an improvement in biodiversity so that the lost ecological value is compensated for. A banking mechanism for offsetting entails a third party providing offsets for developers to purchase, and thus, an offset market emerges. We develop an equilibrium model to analyse the offset markets and apply the model to the Finnish economy and three selected boreal habitats. We examine how the market depends on trading ratios, the presence of an intermediary and the realization of risks associated with uncertainty. An intermediary can facilitate the market participants meeting each other with minimal transaction costs and safeguard against risks. The results show that the size of the offset markets could potentially be quite considerable and providing offsets could be a profitable business for landowners in Finland. The outcome of the restoration investment and a possible time delay between biodiversity losses and gains impact the trading ratios and thus, have a major impact on the market equilibrium. An intermediary may significantly decrease the costs of compensation for developers, provided that it can provide mature offsets when needed.

1. Introduction

Biodiversity is decreasing at an alarming rate. Increasing human population size, land use and consumption cause ecosystem degradation and are among the greatest threats to biodiversity and ecosystem services, as well as to the future of humanity (MEA, 2005). Biodiversity is an essential underlying feature of well-functioning ecosystems that provide numerous ecosystem services: clean water, food, raw materials, nutrient recycling, pollination, climate regulation and recreation, to mention a few. The human use of ecosystem services is growing rapidly; approximately 60% of the ecosystem services evaluated in the Millennium Ecosystem Assessment (MEA) are being degraded or used unsustainably. For instance, half of the provisioning services and 70% of the regulating and cultural services are being degraded (MEA, 2005).

From an economic angle, biodiversity is a public good. Public goods, such as the public ecosystem services that biodiversity sustains, are non-excludable from potential users and non-rival in consumption (Kolstad, 2000). Therefore, a market economy, based on private property and excludability, fails to price and provide them at the desired level (Kolstad, 2000). As a result of this market failure, the price of land does not provide an incentive for developers to consider biodiversity in their decision making. Their businesses result in a negative externality to the environment and lead to habitat loss. Government intervention is needed to save and maintain biodiversity. Current policies focus on the most valuable species and habitats but cannot stop the degradation occurring in less valuable habitats, due to everyday development projects.

One way to address this problem is offsetting the loss of biodiversity caused by development projects (McKenney and Kiesecker, 2010; OECD, 2016; Wende et al., 2018). The basic idea of biodiversity offsetting (also called ecological compensation) is simple: a developer must provide an improvement in degraded habitats elsewhere so that the lost ecological value is compensated for. Usually, compensations are required to guarantee no net loss of biodiversity – sometimes a net gain is set as a more ambitious objective (McKenney and Kiesecker, 2010). Offsets are generated by restoring degraded habitats, creating new

☆ This paper is a part of the Biodiversity Now – Habitat Bank project funded by the University of Helsinki and SITRA, and EKOTEKO project funded by KONE Foundation. The funding is gratefully acknowledged. We would like to thank the two anonymous reviewers for their insightful comments. We also wish to thank Minna Pekkonen, Eeva Primmer and Janne Kotiaho for valuable comments and fruitful discussions.

* Corresponding author.
E-mail address: johanna.a.kangas@helsinki.fi (J. Kangas).

https://doi.org/10.1016/j.ecolecon.2019.01.003
Received 2 May 2018; Received in revised form 3 November 2018; Accepted 4 January 2019
Available online 25 January 2019

habitats and, in some cases, preserving existing valuable ecosystems (OECD, 2016). Developers need not generate the compensations by themselves; they can buy offsets from a third party who provides habitat restoration – often referred as habitat banking (Wissel and Wätzold, 2010). This makes offsetting an economic instrument and allows a market to emerge.

Biodiversity offsetting represents the last resort within a so-called mitigation hierarchy: avoiding, minimizing, restoring and offsetting (Bull et al., 2013; McKenney and Kiesecker, 2010; Wende et al., 2018). Offsetting should compensate only for the residual impacts of development projects after appropriate efforts have been made to avoid damage to ecosystems, to minimize all the unavoidable impacts and to restore biodiversity on-site. Damaging of particularly vulnerable ecosystems and habitats or endangered species should be avoided at all times (BBOP, 2012). Biodiversity offsets should be additional: only those compensation measures that would not have otherwise occurred can be counted as offsetting (McKenney and Kiesecker, 2010; Wende et al., 2018).

From an economic perspective, the no-net-loss objective resembles an emission cap in the trading schemes, because it sets a limit to the biodiversity loss caused by development (McKenney and Kiesecker, 2010; OECD, 2016). Offsetting schemes internalize the external costs of development projects. They establish a counterpart to a 'pollution pays principle', which we call here a 'spoiler pays principle'. Offsetting schemes provide several incentives to conserve biodiversity (Calvet et al., 2015a). Since offsets cause significant costs for developers, they will ex ante reduce impacts on biodiversity. Developers minimize land use and allocate their projects in lands with less valuable habitats. Thus, they intensify the application of the mitigation hierarchy. Developers will also fulfil their offsetting requirement in a cost-effective manner. Lastly, landowners have an incentive to invest in production of offsets, which enables large and expensive conservation projects.

Although biodiversity offsets offer the prospect of achieving further conservation outcomes, they are not a panacea for halting biodiversity loss (Bull et al., 2013; Maron et al., 2016). Ecological risks, as well as the theoretical and practical challenges of offsetting, have been widely discussed in the literature (Calvet et al., 2015b). These problems include difficulties in measuring biodiversity and matching losses and gains (Gamarra et al., 2018; Maron et al., 2012; Quétier and Lavorel, 2011), as well as difficulties to achieve the goal of no net loss (Gardner et al., 2013; Gibbons et al., 2018; Bull et al., 2016a). Time lags between ecological loss and gain, uncertainties and the use of trading ratios and multipliers have also been examined in several publications (e.g. Bull et al., 2016b; Laitila et al., 2014; Moilanen et al., 2009).

Economic analysis of biodiversity offsetting schemes is sparse (Calvet et al., 2015b). Most of the studies have focused on offset production, investments and costs (Coggan et al., 2013b; Fernandez and Karp, 1998; Drechsler and Hartig, 2011), the release of offset credits (BenDor et al., 2014), risks (Levrel et al., 2017), landowners' and intermediaries' incentives (Coggan et al., 2013a; Hartig and Drechsler, 2009), as well as spatial considerations (Bonds and Pompe, 2003; Drechsler and Wätzold, 2009). While providing understanding of many issues impacting market outcomes, these studies have not provided analysis at the market level; their focus was on some specific feature of the market only. Unlike others, Doyle and Yates (2010) analytically and empirically examined the emerging offset market. They found that to achieve no net loss, both economic and ecological processes must be accounted for. Market entry, the quality of restored ecosystems relative to natural ecosystems and especially, the relationship between the ecosystem function and size of the restoration project affect the choice of a trading ratio. When ecological factors are considered, both excess entry and insufficient entry may occur on the offset market.

Previous literature has shown that the design of institutional settings may impact a multitude of market outcomes: equilibrium prices, potential size of the market, transaction costs and realization of risks associated with uncertainty (Vaissière and Levrel, 2015; van Teeffelen et al., 2014; Wissel and Wätzold, 2010). Here, we develop an equilibrium model to analyse the offset markets. We focus on perfect well-functioning markets and examine the impact of trading ratios, the presence of an intermediary and the realization of risks associated with uncertainty. An intermediary, a broker, can facilitate the market participants meeting each other with minimal transaction costs and safeguard against risks. There can be ecological differences between degraded and restored habitats as well as a time delay before gains mature, which calls for the use of trading ratios. The trading is based on the no-net-loss principle.

We apply the analytical model to the Finnish economy and three selected boreal habitats: abundant wetlands, scarce herb-rich forests and expensive and laborious traditional agricultural biotopes. We assume that the supply of offsets comes from habitat restoration and nature management. Data on the areas that are suitable for habitat restoration and associated costs is obtained from many documented sources. For demand, we employ documented predictions on the increase in built-up areas and infrastructure. We use Monte Carlo simulation to examine the impacts of uncertainty on the outcomes of habitat restoration. Estimates of the risks are developed by a survey to specialists on each habitat chosen.

2. Economic Model

2.1. Biodiversity Offset Market Model

Consider a representative landowner making a habitat restoration investment in his/her land. This landowner takes a restoration effort (x) to improve the habitat; x is a continuous decision variable. The restoration effort immediately increases the state of the habitat, which also leads to improvement over time. To formalize this idea, let an ecological function $f(s)$ represent the evolvement of the habitat over time s, and let a restoration function $A(x)$ describe the effect of restoration effort on the evolvement of the habitat. Thus, offset credits from a given habitat area (q) evolve over time by $q = A(x)f(s) - \overline{f}(s)$, where $\overline{f}(s)$ is the evolvement of the habitat in the business-as-usual scenario.

The effort is costly, however. We assume that habitat restoration entails a lump sum investment cost (F) and a unit cost related to effort (w). Let p be the price of offsets. Assume that society has decided that the improvement in biodiversity is determined τ years after the investment and that this improvement can be sold as compensation immediately. In other words, parameter τ represents a point in time when the improvement in the state of the habitat is measured; what the exact number of years τ is, is naturally based on a social agreement. Now, $f(\tau)$ represents the state of the restored habitat at point in time τ and $\overline{f}(\tau)$ represents the state of the habitat in the business-as-usual scenario where the habitat is not restored. We assume that the landowner is risk neutral, i.e. uses the expected values and maximizes profits (π) from habitat restoration in a given land area,

$$max\ \pi = p[A(x)f(\tau) - \overline{f}(\tau)] - wx - F(j), \tag{1}$$

where j is distance and $F'(j) > 0$, indicating that fixed costs increase with distance. This increase in costs arises from the fact that restoration is more expensive in remote areas, for instance, due to higher transport costs for machinery and equipment. Thus, we assume that moving to remoter areas increases the costs and there is a distance that defines the last land parcel restored.

The choice of effort is implicitly determined by

$$\pi_x = pA'(x)f(\tau) - w = 0. \tag{2}$$

Using Eq. (2), the optimal effort is chosen by equating the expected marginal revenue from restoration with the unit cost of the effort. Solving for the effort gives: $x = [pf(\tau) - w]/A'(x)^{-1}$, where $^{-1}$ marks the inverse function. Thus, the choice of effort positively depends on the price of offsets and negatively on the unit price of effort. Eq. (2) holds for any land parcel. How many parcels does the landowner restore? Let π^A denote the return to land in an alternative use. The distance that defines the last land parcel restored is defined by: $j^* : \pi(x^*) = \pi^A$. We use this distance to close the model. This condition, together with Eq. (2), defines the supply of offsets (q^s) as a function of offset prices and costs: $q^s = \int_0^{j^*} [A(x^*(p, w))f(\tau) - \overline{f}(\tau)]dj$. Offset supply is an increasing function of offset price,

$$q_p^s = \int_0^{j^*} A'(x^*(p, w))f(\tau)\frac{\partial x}{\partial p}dj + Af(\tau)\frac{\partial j^*}{\partial p} > 0. \tag{3}$$

We turn next to the need for offsets. A representative developer developing an area for utilization (such as a mine or a production facility) causes biodiversity loss and needs to buy offsets for compensation. How much the developer is willing to pay is dependent on the profitability of the project and the extent of the loss. Thus, the developer obtains utility, i.e. profits, from offsetting as it facilitates the profitable business. Following Doyle and Yates (2010), but using profits instead of utility, we let the developer maximize his quadratic profit function from offsets

$$max\ \pi(q) = aq^d - \left(\frac{b}{2}q^d\right)^2 - pq^d, \tag{4}$$

where a and b are positive constants, q^d is the ecological loss in the habitats requiring offsetting and p is the price of offset credits. Choosing q^d yields $a - bq^d - p = 0$. The marginal profits derived from offsets equal the offset price. This condition results in the demand function $q^d = (a - p)/b$, which is downward-sloping in offset price: $q_p{}^d = -1/b < 0$.

The market equilibrium can now be defined, based on the choices of the representative supplier and demander. We still need to impose the trading ratio, denoted by σ. The trading ratio is calculated between the degraded and restored habitats to set the ecological value of losses and gains equal by adjusting the required amount of restored land. If the ecological value of losses and gains are identical, the trading ratio is equal to 1, and the area of restored habitat is equal to the degraded land area. If the gain is higher than the loss, the trading ratio could be < 1. When the loss is higher than the gain, the ratio should be > 1. In the former case, the restored land area is smaller than degraded land area, and in the latter case it is the other way around. If there is a time delay before generated gains mature, a higher trading ratio is needed. We consider this possibility in the numerical analysis. In the market equilibrium, demand for offsets is dependent on the required trading ratio, so that the equilibrium price is defined where demand meets supply:

$$p^*: q^d(\sigma) = q^s. \tag{5}$$

The trading ratio is calculated case-by-case to adjust for the ecological differences between losses and gains: each development project causes different amount of degradation and each restoration project provides different amount of ecological gains. From Eq. (5), we see that the trading ratios affect the aggregate demand. To illustrate this, consider Fig. 1, where the downward-sloping demand function and the upward-sloping supply function are illustrated. The equilibrium price is determined by their intersection and marked by p^* in both panels of Fig. 1. The total area of compensation sites traded in the market is marked by Q^*. On the market level, we take into account the different case-by-case trading ratios. The aggregate demand is a function of trading ratios employed in individual offsetting cases. Now, if the trading ratios exceed unity ($\sigma > 1$), the aggregate demand curve moves outwards (panel on the left). The developer would be willing to buy the increased amount Q^{**} at price p' but must pay the new equilibrium price p^{**}. If the trading ratios are < 1 ($\sigma < 1$), the aggregate demand curve moves inwards (panel on the right). The developer will then buy a decreased amount of Q^{**} compensation at a lower price p^{**}.

2.2. Parametric Analysis: Scrutinizing Supply

The previous analysis was general, and in this section, we express the model in parametric forms to facilitate the scrutiny of alternative ecologically relevant cases of restoration. This leads to multiple cases of the basic model. We then apply this parametric model numerically to Finnish data.

We use a sigmoid curve to model the evolvement of the habitat,

$$f(\tau) = \frac{L}{1 + e^{-k(\tau - l_0)}}. \tag{6}$$

In Eq. (6), L sets the maximum point of the curve, i.e. the natural state of the habitat, and k determines the slope of the curve. The sigmoid curve provides many advantages for the analysis, as it allows setting the central point l_0, which changes the starting point of the curve and fixes the state of the restored habitat in the beginning, since the restoration effort immediately improves the state of the habitat. The sigmoid curve is first increasing at an increasing rate (convex), then at a decreasing rate (concave). It describes the evolvement of the habitat better than a linear approximation. A sigmoid curve is frequently employed in ecology, for instance, to describe the species-area relationship (e.g. Biedermann, 2003; Tjørve, 2003), the relationship between habitat loss and the probability of extinction (Fahrig, 2001), and the relationship between habitat quality and biological condition (Barbour, 1991). It may be misleading, however, for cases in which rare species emerge in the compensation area over time, making the curve convex, but presumably these cases are rare.

Restoration accelerates the recovery of the degraded habitat towards its natural state. We add the effect of the restoration investment to Eq. (6), using a multiplicative formulation as follows:

$$A(x)f(\tau,\varnothing) - \overline{f}(\tau,\overline{\varnothing}) = (\alpha - \beta x)x\left(\varnothing\frac{1}{1 + e^{-k(\tau - l_0)}}\right) - \overline{\varnothing}\frac{1}{1 + e^{-k(\tau - l_0)}} \tag{7}$$

Parameter $\varnothing$ is added to determine how degraded the state of the habitat is before restoration (in the numerator, $L = 1$, to set a limit to the evolvement to the natural state). Parameter $\overline{\varnothing}$ in turn determines the business-as-usual state of the habitat. We have a quadratic production function $(\alpha - \beta x)x$ with decreasing marginal product to represent the effect of restoration. Parameters α and β are technology parameters.

We consider three illustrative hypothetical cases, all of which are relevant for restoration projects and offsetting markets. They illustrate the differences between the different types of restoration costs and their timing. Here, we provide a parametric analysis of these cases, which is relevant for our numerical calculations in the next section. Case 1 is the simplest: only an up-front investment is required, and no other costs accrue after the first restoration investment. Restoring degraded wetlands provides an example. In case 2, we assume that the costs are periodic: after the initial investment, regular follow-ups are required to maintain the recovery of the habitat. This case is relevant, e.g. for herb-rich forests. Case 3, annual costs, is exemplified with traditional rural biotopes. They require a laborious and expensive investment, because of yearly management. As an extension to this case, we consider a possibility that the landowner must wait for the offset credits to mature before selling.[1]

Case 1. An up-front investment.

Under the above specification, the landowner maximizes profits from restoration investment according to Eq. (7):

[1] We have also examined a case where a percentage of offset credits can be sold in advance, and the remaining credits at a future point in time. The results can be found in Appendix A.

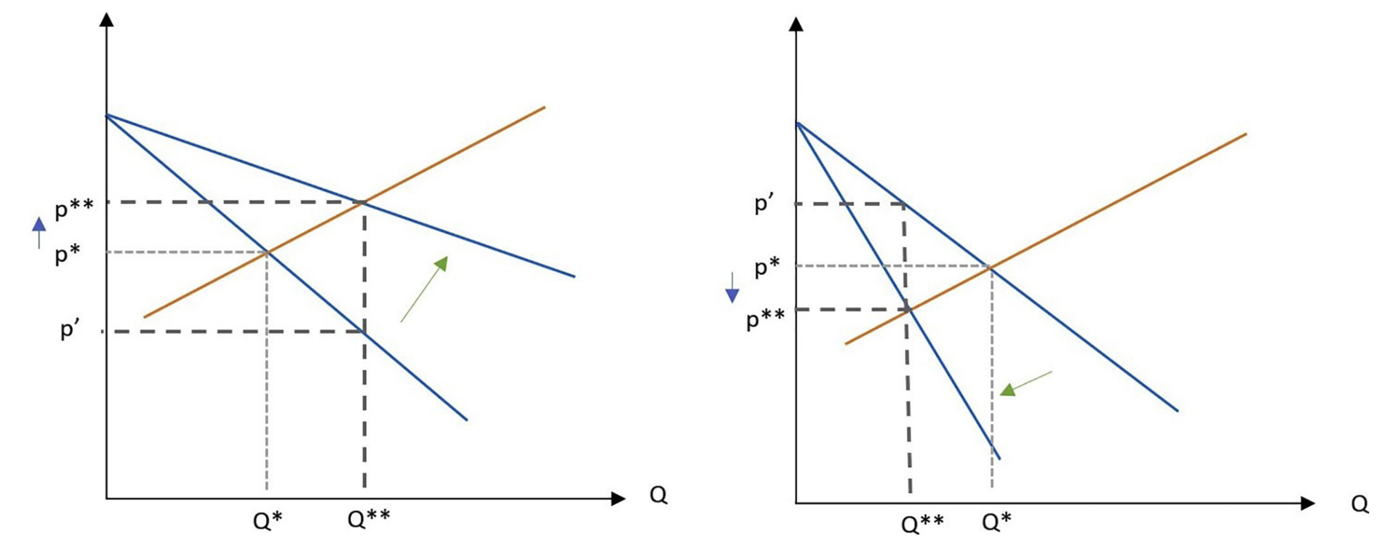

Fig. 1. Impact of a trading ratio on the price of offset credits (p) and the total area of the compensation sites (Q).

$$\pi = p\left[(\alpha - \beta x)x\left(\varnothing\frac{1}{1 + e^{-k(\tau - l_0)}}\right) - \bar{\varnothing}\frac{1}{1 + e^{-k(\tau - l_0)}}\right] - wx - F(j). \tag{8}$$

The optimal restoration effort is defined by the following explicit solution:

$$x^* = \frac{\alpha}{2\beta} - \frac{w}{B}, \tag{9}$$

where $B = p2\beta\varnothing\left(\frac{1}{1 + e^{-k(\tau - l_0)}}\right)$.

Eq. (9) provides the simplest choice of restoration effort. It is characterized by the restoration technology parameters (the first term) and the evolvement of the habitat weighted by the cost-price ratio (the second term).

Comparative statics reveals that the optimal restoration effort is dependent on the technology parameters α and β, the price and variable costs as follows:

$$\frac{\partial x^*}{\partial \alpha} > 0;\ \frac{\partial x^*}{\partial \beta} < 0;\ \frac{\partial x^*}{\partial p} > 0;\ \frac{\partial x^*}{\partial w} < 0.$$

In the restoration function, the technology parameters have opposite impacts: an increase in parameter α increases the optimal effort, and a higher β decreases the optimal effort. An increase in the offset price leads to an increase in the optimal effort, whereas increasing costs decrease it, which means that the choice of effort depends positively on the price of offsets and negatively on the unit price of effort.

The optimal effort depends on the point in time τ and ecological parameters as follows:

$$\frac{\partial x^*}{\partial \varnothing} > 0;\ \frac{\partial x^*}{\partial \tau} > 0;\ \frac{\partial x^*}{\partial l_0} < 0;\ \frac{\partial x^*}{\partial k} > 0.$$

The choice of effort depends positively on ∅, τ and k. Thus, the better the initial state of the habitat, the higher the optimal effort. When the increase in ecological value is measured at a later point in time, the optimal effort increases. An increase in the slope of the curve increases the optimal effort as well. Parameter l_0 has a negative impact on the optimal effort. It fixes the starting point of the curve, since the restoration effort immediately improves the state of the habitat. Thus, the smaller this impact, the higher the optimal effort.

The point in time when the improvement in the state of the habitat is measured impacts credits supplied in the following manner:

$$q = A(x)f(\tau,\varnothing) - \bar{f}(\tau,\bar{\varnothing})$$

$$\frac{\partial q}{\partial \tau} = A'(x)\frac{\partial x}{\partial \tau}f'(\tau,\varnothing) - \bar{f}'(\tau,\bar{\varnothing}).$$

Thus, if $\bar{f}(\tau,\bar{\varnothing})$ is decreasing, measuring the increase in ecological value at a later point in time increases the supplied amount of credits. However, if $\bar{f}(\tau,\bar{\varnothing})$ is increasing, i.e. the state of the habitat improves also in the business-as-usual scenario in which the habitat is not restored or managed, the impact of τ is ambiguous.

Case 2. Periodic costs.

In case 2, we assume that after the initial investment (*F*), additional restoration is needed, which occurs at time *m*. Thus, the profit function can be expressed as follows:

$$\pi = p\left[(\alpha - \beta x)x\left(\varnothing\frac{1}{1 + e^{-k(\tau - l_0)}}\right) - \bar{\varnothing}\frac{1}{1 + e^{-k(\tau - l_0)}}\right] - wx(1 + r)^{-m} - F(j). \tag{10}$$

The optimal effort is defined by

$$x^* = \frac{\alpha}{2\beta} - \frac{w(1 + r)^{-m}}{B}. \tag{11}$$

Comparing Eq. (11) with Eq. (9) reveals how discounting impacts the outcome. The difference lies in the discounting factors $(1 + r)^{-m}$ multiplying the term w/B. It means that the further in time the future costs of effort occur (higher *m*), the higher the restoration effort. Qualitatively, comparative statics remains the same. The impact of the interest rate is positive: an increase in the interest rate increases the optimal effort.

$$\frac{\partial x^*}{\partial r} > 0.$$

Case 3. Costs accrue annually.

Suppose, that there is a habitat that needs continuous management and thus, restoration costs occur every year. We use annuity to discount the costs, and the profit function is given by

$$\pi = p\left[(\alpha - \beta x)x\left(\varnothing\frac{1}{1 + e^{-k(\tau - l_0)}}\right) - \bar{\varnothing}\frac{1}{1 + e^{-k(\tau - l_0)}}\right] - wx\left(\frac{1 - (1 + r)^{-m}}{r}\right) - F(j). \tag{12}$$

The optimal effort is now defined by

$$x^* = \frac{\alpha}{2\beta} - \frac{[1-(1+r)^{-m}]}{r}\frac{w}{B}. \tag{13}$$

Further, we examine an option where the revenue the landowner receives is obtained at a future point of time, denoted by *n,* the profit function is

$$\pi = p\left[(\alpha - \beta x)x\left(\varnothing\frac{1}{1+e^{-k(\tau-l_0)}}\right) - \bar{\varnothing}\frac{1}{1+e^{-k(\tau-l_0)}}\right](1+r)^{-n} - wx\left(\frac{1-(1+r)^{-m}}{r}\right) - F(j). \tag{14}$$

The optimal effort in this case is defined by

$$x^* = \frac{\alpha}{2\beta} - \frac{[1-(1+r)^{-m}]}{r(1+r)^{-n}}\frac{w}{B}. \tag{15}$$

Again, the difference in comparison with the benchmark Eq. (9) is defined by the term $[1-(1+r)^{-m}]/r(1+r)^{-n}$. Irrespective of the sizes of n and m, $[1-(1+r)^{-m}]/r(1+r)^{-n} > (1+r)^{-m}/1$. Thus, relative to the previous case, the latter negative term increases, and when costs occur annually, the optimal effort decreases. In this case, the effect of the discount rate is ambiguous. Otherwise, comparative statics remains the same.

Our formal analysis of investments in producing offsets and the market is now complete. We next apply the model to empirical cases.

3. Data

3.1. Habitats and Restoration Measures

We apply the model to Finnish data with three habitat types: pine mires, herb-rich forests and traditional rural biotopes. The habitats are representative of the environment in Finland and highlight the differences in restoration costs and timing of the investment. We utilize the results of the working group on improving the status of habitats in Finland (Kotiaho et al., 2015) for the valuation of the ecological state of each habitat, habitat-specific restoration and nature management measures and the cost estimations of the investments.

The valuation of the ecological state of each habitat is based on habitat-specific structural characteristics, which are weighted according to their importance for biodiversity. Following the ELITE method presented in Kotiaho et al. (2015), the ecological state can range from 0 to 1, where 1 is equivalent to a habitat in its natural state, or in the case of rural biotopes and herb-rich forests, the target state of the habitat. Eq. (16) is used to calculate the state of the habitat in its current state (Kotiaho et al., 2015):

$$R = \prod_{n=1}^{N}\left(1 - L_n\left(1 - \frac{n_{curr}}{n_{ref}}\right)\right). \tag{16}$$

Here, R is the current state of the habitat, N the number of structural characteristics and L_n the weight indicating the importance of each characteristic to biodiversity. The weight is a percentage by which the state of the habitat degrades if that factor is completely lost. The terms n_{cur} and n_{ref} are the current state and the natural state of characteristic n.

Landowners supply biodiversity offsets for compensation by performing restoration and nature management measures, thus resulting in additional biodiversity gains that would not otherwise occur. Restoration includes measures that initiate or accelerate the recovery of an ecosystem towards its original natural state (Kotiaho et al., 2015). Nature management refers to measures aimed at maintaining a habitat in certain phases of succession that are the most crucial to biodiversity (Similä and Junninen, 2011). There is a wide array of methods that aim to enhance ecosystem recovery, and next we describe them in further detail for each habitat type selected. Since it is important to ensure the long-term existence of the offsets, we assume that compensation areas are established as permanent conservation areas or preserved in another legally binding fashion.

3.1.1. Pine Mires

As the first habitat type, we focus on the restoration of oligotrophic pine mires. A mire is geologically defined as an area with a peat layer of at least 30 cm, but the area covered by mires is considerably larger in biological terms (Raunio et al., 2008). Pine mires are usually nutrient poor, with a thick peat layer. Peatland drainage for forestry and peat harvesting are the most important causes of threats to mire species (Rassi et al., 2010). Construction of infrastructure and groundwater abstraction have also deteriorated mires. Currently, only about 40% of the original area of mires has been left undrained in Finland. Approximately half of all mire habitat types are classified as threatened. There are 4.7 million ha of pine mires in Finland, of which 2.8 million ha are drained (Kotiaho et al., 2015). We consider a pine mire that is drained, but peat harvesting and forestry are unprofitable, which makes it a good area for restoration.

Hydrology is the most important factor affecting the state of mire ecosystems. Drainage and other land use disturb the hydrology by blocking water flow to the mire or increasing the outflow of water. Drainage especially lowers the water level in mires and increases the number of trees. Thus, mire species are replaced with species adapted to dryer and shadier environments. The ecological state of pine mires is presented in Table 1. Drawing on Eq. (16), the current state is estimated, based on tree stand and hydrology. Hydrology is estimated roughly on a percentage basis: 100% represents hydrology in the natural state and 0% a completely degraded hydrology in which natural water flow is nonexistent. Hydrology is given more weight, due to its high importance to the state of mire ecosystems. The current state of pine mires is on average 0.32 (Kotiaho et al., 2015).

Restoration measures consist of filling drains and removing tree stands to an amount consistent with a pine mire in a natural state (Kotiaho et al., 2015). When the drains are filled, the water level is expected to reach its natural state and the flow of water recovers. The recovering hydrology is a precondition for the recovery of structure and functions of a mire ecosystem. Removing trees will restore the openness of the mire and partly affect the water levels as well.

3.1.2. Herb-rich Forests

The second habitat type we consider is herb-rich forests, which are Finland's most lush and species-rich forest type, and usually support several tree species (Raunio et al., 2008). In Finland, almost half of the threatened forest species and > 40% of all red-listed forest species live primarily in herb-rich forests (Rassi et al., 2010). Various types of herb-rich forests are also among the most threatened forest habitat types (Raunio et al., 2008).

The amount of decaying wood is one of the most important factors affecting the diversity of forest species (Rassi et al., 2010). Forest management activities, changes in the tree species composition, reduction of old-growth forests and the decreasing numbers of large trees are also significant threats (Raunio et al., 2008). Forestry and the lack of natural disturbance dynamics have long reduced the variety of tree species in forest stands. Forests are even aged, they lack the diversity of early succession forests and old-growth forests are rare. In recent years, the use of wood provided by forests has been intensified in Finland, e.g. by shortening the felling cycle and collecting logging residue and

Table 1
Structural characteristics of pine mires.

	Tree stand	Hydrology	State of the habitat
Natural state	20 m^3/ha	100	1.0
Weight	0.1	0.95	
Current state on average	30 m^3/ha	30	0.32

Table 2
Structural characteristics of herb-rich forests.

	Number of large trees	Amount of decaying wood	Volume of broad-leaved trees	State of the habitat
Target state	30 m^3/ha	100 m^3/ha	100 m^3/ha	1.0
Weight	0.4	0.4	0.6	
Current state on average	10.1 m^3/ha	7 m^3/ha	92 m^3/ha	0.44

stumps for biofuel, which will further decrease the amount of decaying wood remaining in forests (Rassi et al., 2010).

We refer to a target state instead of a natural state, since with management, herb-rich forests are maintained in certain phases of succession to prevent the natural proliferation of spruce, which negatively impacts the diversity of herb-rich forest species. Three structural characteristics that indicate the degradation of herb-rich forests are presented in Table 2: the number of large trees (with a diameter of at least 40 cm), the amount of decaying wood and the volume of broad-leaved trees (Kotiaho et al., 2015). These factors are crucial to the diversity of forest species and forest habitats. Large trees serve as important perches for predatory birds and epiphytes. Large trees also produce important large-sized decaying wood. Broad-leaved trees are especially crucial to biodiversity in herb-rich forests and, thus, their volume is given the greatest weight. Currently, the state of herb-rich forests is on average 0.44.

Herb-rich forests are restored using nature management measures that aim to establish forests dominated by broad-leaved trees, with a diverse tree stand structure, decaying wood and large trees (Kotiaho et al., 2015). Although takeover by spruce is part of the natural succession, it negatively impacts the diversity of herb-rich forest species. Thus, the objective of nature management is not to achieve a forest in its natural state, but to maximize biodiversity in the habitat. The nature management measures include reducing the number of spruce and managing the forest, regularly if needed, to prevent the natural proliferation of spruce, to increase the proportion of broad-leaved trees and to secure variation in the tree stand structure. Formation of decaying wood can also be promoted, where appropriate.

3.1.3. *Rural Biotopes*

Finally, we examine traditional rural biotopes, which include various open, dry, mesic and moist grasslands resulting from grazing or mowing, as well as wooded pastures and meadows. All traditional rural biotopes are classified as critically endangered or endangered in Finland (Rassi et al., 2010). Their area has declined by > 90% since the 1940s, with their quality considerably deteriorated. > 20% of red-listed species have rural biotopes as their main habitat. The most significant threat to rural biotope species is the overgrowth of meadows and other open habitats. Open habitats have closed up, due to changes in farming and pasturing practices; the number of small- or intermediate-sized farms has declined, and in many areas traditional grazing and mowing have either ceased or decreased considerably. Conservation of traditional rural biotopes requires continuous management. Currently, 30,000 ha are managed, while the minimum target for ensuring the conservation of the most important species is 60,000 ha (Kemppainen and Lehtomaa, 2009).

Again, we refer to a target state instead of a natural state, since rural biotopes are shaped and maintained by human activities and lose the biodiversity characteristics typical of these habitats without constant or repeated management. The structural characteristics needed to estimate the ecological state of rural biotopes are presented in Table 3: vegetation, the openness of the habitat and the history of soil cultivation (Kotiaho et al., 2015). Vegetation refers to the condition of the field layer. Overgrowth, eutrophication and poor management disturb the plant species typical of rural biotopes. Openness of the habitat refers to the fact that rural biotopes are typically open grasslands, pastures and meadows with diverse field layers and few trees. The increasing numbers of trees and shrubs reduce the typical openness of the habitat and replace the species adapted to open ecosystems. The history of soil cultivation affects the plant species composition in the habitat. Fertilization and turning the habitat to intensive agricultural use are harmful, because soil cultivation alters the structure of the soil and thus affects the plant species. The habitat is in its target state when all factors are in 100% condition. The current state of rural biotopes is on average 0.06. Thus, the state of traditional rural biotopes is weakest in comparison with the other habitat types considered in this study.

Table 3
Structural characteristics of traditional rural biotopes.

	Vegetation	Openness of the habitat	History of soil cultivation	State of the habitat
Target state	100%	100%	100%	1.0
Weight	0.85	0.75	0.95	
Current state on average	10%	20%	60%	0.06

Traditional rural biotopes require repeated management measures to maintain the habitat conditions preferred (Kotiaho et al., 2015). If a rural biotope has been unattended for a prolonged time, it requires a thorough renovation that includes thinnings, removing coppices and young trees and mowing unfavourable vegetation. Thereafter, the biotope is managed annually to prevent overgrowth and to maintain open areas. The repeated management measures include grazing and mowing, which maintain the habitat characteristics preferred and enable the survival of the fauna and flora typical of rural biotopes. Thus, the species composition characteristic of rural biotopes is maintained.

3.2. *Uncertainties of Habitat Recovery*

There is uncertainty in how different habitats respond to restoration and management. To map the scope of these uncertainties, we designed and conducted an expert survey. The objective was to estimate how the habitats would develop without restoration and/or nature management measures compared with a business-as-usual scenario, and how the outcomes of restoration and nature management measures vary under uncertainties. The survey was conducted for each habitat type separately, and the respondents were experts specialized in the ecology of the habitat in question. Ten respondents received the surveys regarding pine mires and rural biotopes, while the survey for herb-rich forests was sent to nine respondents. The experts represented the up-to-date understanding of these habitat types in Finland.

In the introduction of the survey, both the habitat type and the specific restoration measures were described as mentioned in this study. We first asked the respondents to estimate, based on their best knowledge, how the habitat would develop without any restoration measures in 100 years and in 200 years. The respondents were also asked to give estimates for the most likely value, as well as the minimum and maximum values, for the possible outcomes of habitat restoration after 100 years. The respondents were also asked to estimate the probabilities for each value. Finally, the respondents estimated the probability that the restored habitat would reach a natural state after 200 years. In addition, after each question, the respondents were asked to give a confidence level for their answer on a Likert scale 1–5 (1 = completely uncertain, 5 = completely certain). The responses are

Table 4
Weighted averages of survey responses.

	Current state	Restored, 100 years			Business-as-usual	
		Lower bound	Most likely	Upper bound	100 years	200 years
Pine mires	0.32	0.63	0.79	0.89	0.45	0.6
Herb-rich forests	0.44	0.68	0.84	0.95	0.21	0.18
Rural biotopes	0.06	0.56	0.85	0.94	0.04	0.005

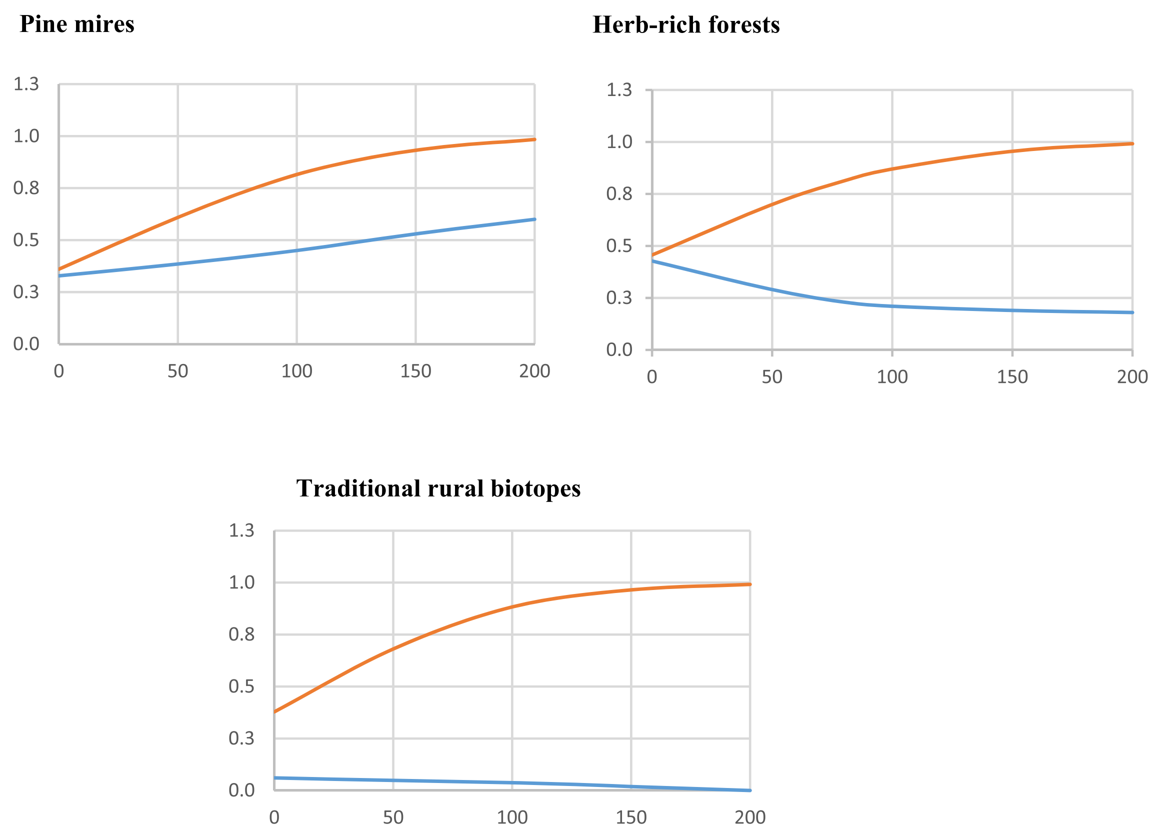

Fig. 2. Evolvement of the ecological state of the habitats. X-axis represents time and y-axis represents the ecological value of the habitat.

presented in Table 4 and Fig. 2 below. The averages are weighted based on the confidence levels given. We present the questionnaire as well as the results and a statistic summary in Appendix B.

The responses show that the state of pine mires gradually develops towards the natural state without restoration (Table 4). Since we assume here that peat harvesting and forestry are unprofitable, drainage maintenance would stop and the hydrology would slowly start to recover. However, filling the drains and removing the trees would cause the mire to recover more rapidly towards its natural state. The state of herb-rich forests would decrease significantly without nature management measures, mostly due to forestry and the lack of natural disturbance dynamics. Nature management measures would restore the structural characteristics of herb-rich forests, above all by removing spruce and leaving broad-leaved trees standing. Currently, traditional rural biotopes are in a highly degraded state, and without management their state would gradually fall to zero. Thinnings, removing coppices and mowing unfavourable vegetation would restore the openness of the habitat, and annual management would cause the habitat to recover towards its target state. The respondents were optimistic about succeeding in restoration; their estimate for the likelihood of the habitat reaching its natural state in 200 years is on average 92% in pine mires, 94% in herb-rich forests and 92% in rural biotopes.

We use Eq. (6) and Table 4 to illustrate the evolvement of habitats in Fig. 2 over a 200-year time period. The upper lines represent the most likely case under habitat restoration, based on the expert estimates, which show that the habitats gradually approach their natural state. The lower graphs indicate how the habitats were predicted to evolve over time if they were not restored or managed. They illustrate the above-described finding that the state of herb-rich forests and traditional rural biotopes would degrade over time and decline to near-zero.

Pine mires differ from herb-rich forests and traditional rural biotopes in that their state develops towards the natural state without restoration if they are left intact. Thus, the improvement in the ecological state, which would be achieved with restoration, is considerably lower in pine mires than in the other habitats in this study. The increase in the state of the habitat is 0.34 units (recall that the ecological value is scaled between 0 and 1) after 100 years and 0.40 units after 200 years, whereas in traditional rural biotopes the same figures are 0.81 and 1.0. Since the traded offset credits are calculated by an ecological index value representing an increase in the ecological state in a restored land area, this suggests that there would be fewer offset credits for sale per hectare in pine mires than in the other habitat types.

3.3. Parameters in the Simulation Model

In the previous sections, we introduced the habitat types selected for the application of the model, measures to restore and manage them, as well as the evolvement of the habitats with and without restoration and

Table 5
Parameters in Eq. (7).

		Pine mires	Herb-rich forests	Rural biotopes
Technology parameter	α	3.9	3.9	3.9
Technology parameter	β	2.1	2.1	2.1
Degraded state of the habitat	Φ	0.62	0.565	0.77
Central point of the curve	l_0	30	10	20
Slope of the curve	k	0.02	0.02	0.025
Point in time	τ	50	50	50

management. Now, we represent the parameters in the model, which are scaled in accordance with data on the evolvement of the habitats from the expert survey, and the costs of restoration and nature management (Kotiaho et al., 2015). Recall Eq. (7) representing the evolvement of the habitat with restoration effort:

$$A(x)f(\tau\varnothing) = (\alpha - \beta x)x\left(\varnothing\frac{1}{1 + e^{-k(\tau - l_0)}}\right) - \bar{\varnothing}\frac{1}{1 + e^{-k(\tau - l_0)}}.$$

Table 5 includes the parameters in the empirical application of the restoration model. Pine mires are an example of case 1, herb-rich forests represent case 2 and traditional rural biotopes case 3.

The technology parameters were scaled so that the effort varies between 0 and 1 and full effort would cause the habitat to evolve optimally towards its natural or target state. The degraded state of the habitat was scaled so that it corresponds to the data represented in Tables 1–3. The central point and slope of the curve are scaled to correspond to the data on the evolvement of the restored and managed habitats, derived from the survey.

Now we turn to the economic parameters, which are found in Table 7. For cost estimates, we utilize data from Kotiaho et al. (2015). Pine mires are usually restored once so there are only up-front costs. The price of timber is 30 €/m^3 in the calculations. A fixed cost, the total cost of conservation, is 1000 €/ha and includes the value of the current tree stand (20 m^3) at 600 €/ha and the value of land at 100 €/ha. An administrative cost of 300 €/ha, associated with establishing a conservation area, is also added. A variable cost, the restoration investment, is 1400 €/ha. First, it includes the costs of removing the trees (10 m^3) at 1000 €/ha and 300 €/ha revenue from selling timber. Second, the cost of filling the drains with an excavator is 500 €/ha. A planning cost of 200 €/ha is also added.

Herb-rich forests require a larger fixed investment in nature management in the beginning and follow-ups (a variable cost of 150 €/ha) after the initial management investment (Kotiaho et al., 2015), we assume that they are performed 20 and 40 years later. The nature management measures and their costs vary widely among sites, depending on the state and age of the tree stand. Clearing a stand of spruce saplings can be extremely costly, whereas the removal of mature spruce can yield substantial sales revenue. We update the cost estimations presented in Kotiaho et al. (2015) for this analysis as they used a maximum clearing cost as a basis and we want to take into account the wide variety of costs related to the management of herb-rich forests.

First, the fixed cost of conservation is approximately 7400 €/ha. We calculate it as a bare land value for managed spruce forest land and add an administrative cost of 20% (Kotiaho et al., 2015). Second, we calculate the costs of nature management measures separately for a site dominated by spruce and for one dominated by broad-leaved trees. The age of the tree stand is also taken into account: costs are different for saplings, young tree stands and mature tree stands. For the cost of nature management measures *F*, consider Eq. (17) and Fig. 3. Parameters used in the calculations are presented in Table 6. Prices for saw timber and pulpwood are from the statistics of Natural Resources Institute Finland (LUKE, 2016). The vertical axis represents the growth in forest value and the horizontal axis represents time. Spruces are removed at time *t'*. The optimal rotation time is *T**. Thus, the landowner faces a cost for not clear cutting at time *T**, but instead removing only spruces at time *t'*. The term $h(T^*)$ represents the revenue from clear cutting at time *T** and $h(t')$ the revenue from harvesting the spruces at time *t'*. The fixed costs are discounted by a factor $(1 + r)^{T^* - t'}$. We add a planning cost of 150 €/ha.

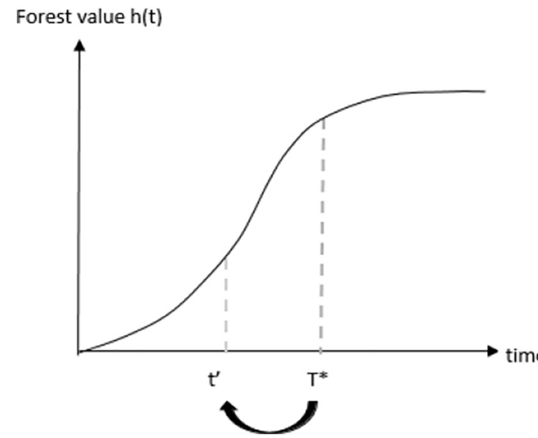

Fig. 3. Costs of removing spruce at time t' versus time T*.

Table 6
Parameters for calculating the management costs in herb-rich forests.

	Broad-leaved trees dominate	Spruces dominate
Interest rate	0.03	0.03
Price, saw timber (€/m^3)	40	50
Price, pulpwood (€/m^3)	15	17

$$F = p[h(T^*)(1 + r)^{t' - T^*} - h(t')] + 150 \tag{17}$$

In rural biotopes, fixed costs include the cost of conservation, 4987 €/ha, which is based on the value of land (Kotiaho et al., 2015). The fixed costs also include the cost of clearance of dense vegetation is needed before grazing, which is estimated to be 1862 €/ha. Variable costs, 875 €/ha per year, include the costs of annual grazing and the clearing of vegetation, if necessary. A guidance cost of 5% is added to the annual cost. The rural biotopes are assumed to be managed for 50 years.

3.4. Estimates for Supply and Demand

The estimates for potential supply and demand are provided in Table 8. We define the potential supply of offsets from each selected habitat type as the area suitable for restoration in Finland. The estimates of these land areas are based on expert assessments and literature (Kemppainen and Lehtomaa, 2009; Kotiaho et al., 2015). We estimate the demand for offsets drawing on the predictions of future land use change. Tiitu et al. (2015) predicted changes in the area of settlements in Finland for the time period 2013–2040. To estimate potential demand, we utilize their predictions of built-up areas (such as residential areas, industrial and commercial complexes, areas for sports and

Table 7
Parameters in the profit functions.

		Pine mires	Herb-rich forests	Rural biotopes
Variable cost (€)	w	1400	150	918.75
Fixed cost (€)	F	1000	9264	6849
Interest rate (real)	r	–	0.03	0.03
Timing of variable costs (years)	m	–	20 & 40	50

Table 8
Total area of each habitat in Finland, areas suitable for restoration and land-use pressure, in hectares to the year 2040.

	Total area in Finland	Restorable area	Land-use pressure
Pine mires	193,000	193,000	33,000
Herb-rich forests	377,600	264,000	2500
Rural biotopes	100,000	30,000	3300

Table 9
Market equilibrium: no time delay, trading ratio 1.

	Price, €/ha	Profits, €/ha	Compensation sites in total, ha
Pine mires	8150	117	31,393
Herb-rich forests	12,000	363	2000
Rural biotopes	29,559	557	3197

Table 10
Market equilibrium: 15-year time delay, trading ratio 1.6.

	Price, €/ha	Profits, €/ha	Compensation sites in total, ha
Pine mires	8471	206	46,888
Herb-rich forests	12,500	568	3000
Rural biotopes	30,088	919	4961

Table 11
Total size of the hypothetical offset market.

Trading ratio	Total size, M€		Total area, ha	
	1	1.6	1	1.6
Pine mires	256	397	31,393	46,888
Herb-rich forests	24	38	2000	3000
Rural biotopes	95	150	3197	4961
Total	374	584	36,590	54,849

Table 12
Trading ratios when the outcomes of restoration vary (t = 15 years, interest rate 3%).

	Loss	Gain	Trading ratio	
			No time delay	Time delay 15 years
Pine mires	0.5	0.2	2.5	4
Herb-rich forests	0.5	0.35	1.5	2.3
Rural biotopes	0.5	0.6	0.8	1.3

Table 13
Market equilibrium: the outcomes of restoration vary, no time delay.

	Trading ratio	Price, €/ha	Profits, €/ha	Compensation sites in total, ha
Pine mires	2.5	8880	320	66,667
Herb-rich forests	1.5	12,421	535	2842
Rural biotopes	0.8	29,375	432	2585

Table 14
Market equilibrium: the outcomes of restoration vary, a 15-year time delay.

	Trading ratio	Price, €/ha	Profits, €/ha	Compensation sites in total, ha
Pine mire	4	9420	471	92,749
Herb-rich forests	2.3	13,010	776	4019
Rural biotopes	1.3	29,828	741	4093

recreation) and infrastructure (including roads, airports, extraction sites, ports, dump sites). The report provides estimates for how many hectares of land in each habitat type will turn into built-up areas or infrastructure. We add the leakage of development impacts outside the area (20%) and have also taken into account future peatland use (1000 ha/year), based on a report by Leinonen (2010), and the objectives of the Finnish National Energy and Climate Strategy (Kansallinen energia- ja ilmastostrategia, 2013).

Direct land use changes and other activities also cause indirect impacts, leading to decrease in habitat quality. Accounting for these will considerably increase the size of land areas under pressure. The magnitude of these impacts is very difficult to estimate. We include an increase of 100 ha/year in pine mires and 50 ha/year in herb-rich forests and rural biotopes. The location of the demand curve is approximated based on the estimations of land use pressure for each habitat (Table 8), so that the market would cover the demand rising from land use.

3.5. Uncertainty and Monte Carlo Simulations

Recall Section 3.3, in which uncertainty concerning the success of restoration was examined. Next, we take the estimated uncertainties into account in the model. Monte Carlo simulation allows us to examine the possible outcomes of habitat restoration when its success may vary. Monte Carlo simulation uses probability distributions to capture the uncertainty of the variables under scrutiny and therefore is well suited for the analysis of uncertainty. The simulation was performed in accordance with the results of the survey (for combining an expert survey and Monte Carlo simulation, see Bamber and Aspinall, 2013). The results suggest the use of a triangular probability distribution in the simulation. The minimum, most likely and maximum values are defined; values around the most likely figure are more likely to occur. The program recalculates the results, each time using a different value. The values are selected at random from the input probability distribution thousands of times. As a result, it produces distributions of possible outcome values, explaining what could happen and how likely the outcome is.

Variation in the evolvement of restored habitats over a 200-year timespan is shown in Fig. 4. When accounting for uncertainty, the improvement in the state of the restored habitats in rural biotopes and herb-rich forests is still striking, since their state would significantly degrade without restoration. The state of pine mires would improve even without restoration. However, with restoration the improvement is more rapid and recovery closer to the natural state more likely.

Fifty years after the restoration, 90% of the restoration outcomes are between 0.48 and 0.65 in pine mires, 0.59 and 0.73 in herb-rich forests and 0.48 and 0.69 in rural biotopes. The minimum values are 0.44 in pine mires, 0.55 in herb-rich forests and 0.42 in rural biotopes. Thus, the spread of uncertainty is widest in rural biotopes. In all the habitats selected, interestingly, the most likely values| are closer to the upper than the lower bounds. The lower bounds are still higher in ecological value than the state without restoration or management measures.

Furthermore, it is useful to know how many restoration sites will fail to provide sufficient biodiversity gains to be sold as compensation. We assume that if the restoration outcome is < 90% of the expected value, restoration has failed. Thus, 20% of the pine mire sites and 10% of the herb-rich forest sites would not be saleable. Since the improvement achieved with restoration is so substantial in rural biotopes, we assume that the restoration outcome must be at least 85% of the expected value to be accepted as compensation. Thus, 10% of the rural biotope sites would not be saleable. These figures are later taken into account in a risk assessment.

Table 15
Impact of τ on trading ratios, no time delay.

	Loss	Gain ($\tau = 25$)	Gain ($\tau = 50$)	Gain ($\tau = 100$)	σ ($\tau = 25$)	σ ($\tau = 50$)	σ ($\tau = 100$)
Pine mires	0.5	0.1	0.2	0.3	5	2.5	1.7
Herb-rich forests	0.5	0.2	0.35	0.6	2.5	1.5	0.8
Rural biotopes	0.5	0.45	0.6	0.8	1.1	0.8	0.6

Table 16
Market equilibrium: $\tau = 25$, no time delay.

	Trading ratio	Price €/ha	Profits €/ha	Compensation sites in total, ha
Pine mires	5	9706	552	106,589
Herb-rich forests	2.5	13,143	831	4286
Rural biotopes	1.1	27,858	751	3860

4. Perfectly Functioning Offset Markets, With and Without Time Delay

Next, we apply the analytical model to examine the offset markets numerically. We especially want to examine how the market equilibrium – the prices and quantities traded – depends on the trading ratios. The theoretical analysis is done assuming a perfect and well-functioning market where mature offsets are available when needed. This type of market may be possible when an intermediary, a broker, works in the market. It aids the demanders and suppliers in meeting each other with minimal transaction costs. An intermediary can also safeguard against the risks associated with restoration by buying restored habitats beforehand, so that whenever degrading of a habitat occurs, the intermediary is able to supply a verified restored habitat for compensation. If the intermediary is not present in the market, possible time delays between biodiversity losses and gains must be accounted for either by increasing trading ratios or by landowners who must wait for the credits to mature before selling.

In the benchmark case we set the trading ratio to unity. If there are no mature offsets when losses occur, a time delay exists between the losses and gains. We take this delay into account by discounting the improvement in ecological value to the present. We use the following equation to calculate the trading ratio to match the ecological value of the loss and the discounted gain:

$$\sigma = \frac{q^d}{q^s(1 + r)^{-t}}, \tag{18}$$

We assume that it would need t = 15 years to ensure that habitat restoration has succeeded as expected and offsets mature and use discount rate r = 3%. Using Eq. (18), a time delay of 15 years yields an increase in the trading ratio to 1.6. We compare this to an option where the landowner waits for the credits to mature and gets revenue only 15 years after the investment. In the next section, the analysis is complicated by taking into account the varying outcomes of restoration in different habitats.

Recall that in Eq. (3), we chose to use distance j to close the model. Thus, the aggregate supply of offsets from each habitat is derived by assuming that fixed costs increase when distance j increases, since remoter sites are more difficult to reach, and that distance defines the last land parcel restored. For simplicity, the restoration costs are assumed to be homogenous in other respects; only their fixed costs differ.

Table 18
Market equilibrium with risk premium.

	Risk premium €/ha	Buyer price €/ha	Profits €/ha	Compensation sites in total, ha
Pine mires	1630	9587	64	22,089
Herb-rich forests	1200	13,067	308	1733
Rural biotopes	5912	35,163	348	2170

Table 19
Market equilibrium with higher risk premiums.

	Risk premium €/ha	Buyer price €/ha	Profits €/ha	Compensation sites in total, ha
Pine mires	2852	10,665	24	15,112
Herb-rich forests	3000	14,667	226	1333
Rural biotopes	8868	37,965	243	1656

4.1. Benchmark for Market Equilibrium

Table 9 presents the benchmark with no time delay and the trading ratio equal to 1. The results are presented in terms of equilibrium prices, profits and total area of the compensation sites.

The equilibrium prices and quantities vary widely, depending on the habitat in question. Offset credits in pine mires are the cheapest, and the restored land areas are the largest, because they require the least costly investment and the land use pressure is strongest. Herb-rich forests are 50% and traditional rural biotopes four times more expensive than pine mires. Landowners' net profits are slightly over 100 €/ha for pine mires and approximately 400 and 600 €/ha for other habitat types. The costs to companies needing offsets in these habitats are in total approximately 370 million €.

If we add time delay, the gains must be discounted to the present. With a 15-year delay and 3% discount rate, the trading ratio increases from 1 to 1.6. Features of the market equilibrium are presented in Table 10.

Relative to Table 9, both the equilibrium prices and restored land areas increase. A higher trading ratio means that more land is needed to compensate for the biodiversity losses: the increase is approximately 50%. The increase in prices is moderate: 2–4%. The profits for landowners also increase. The costs to companies needing offsets increase to approximately 580 million €.

An alternative way to account for the time delay is that the landowner must wait for offsets to mature before selling. Instead of the

Table 17
Market equilibrium with an intermediary fee, trading ratio 1.

	Intermediary's fee, €/ha	Buyer price, €/ha	Profits, €/ha	Compensation sites in total, ha
Pine mires	407	8509	104	29,067
Herb-rich forests	600	12,533	336	1867
Rural biotopes	1478	30,960	505	2941

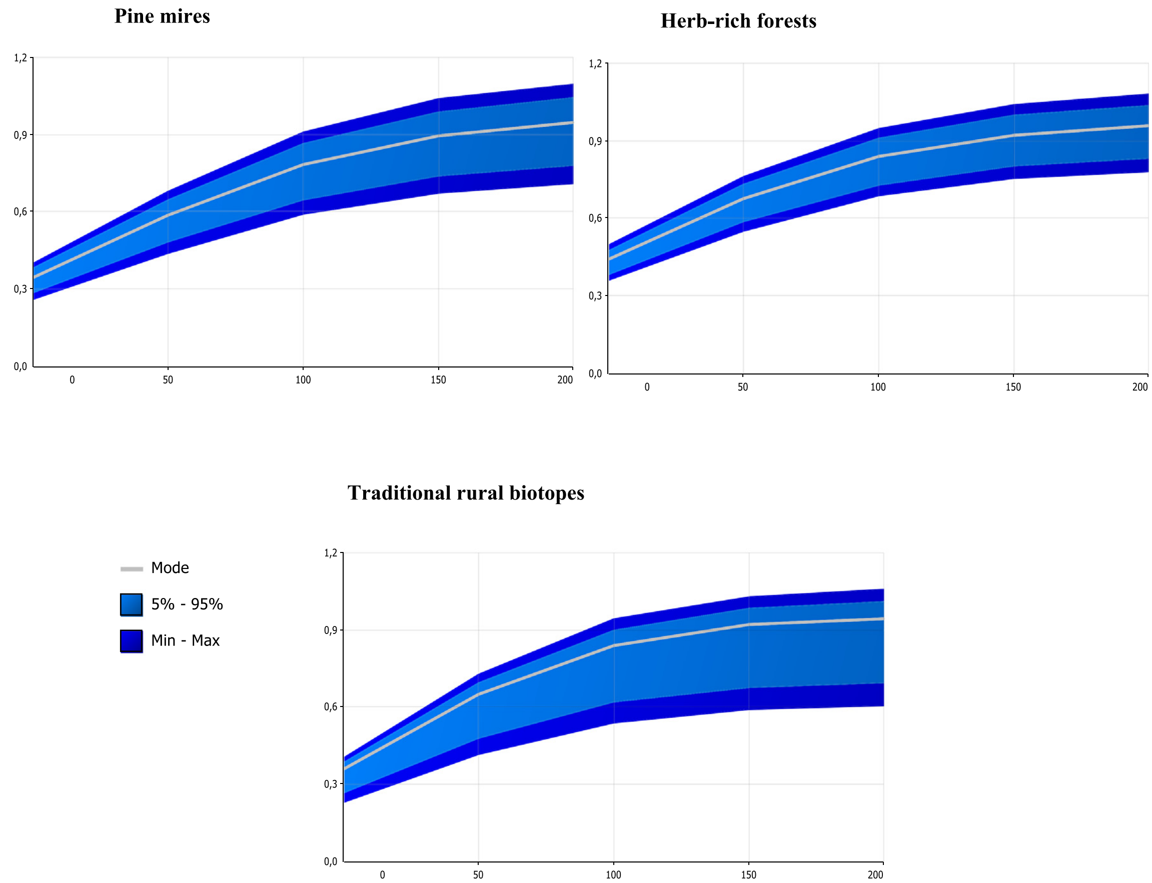

Fig. 4. Variation in the evolvement of restored habitats.

developer buying credits before they are mature and compensating for that by buying more (Table 10), the landowner bears the cost of waiting for the credits to mature and does not get revenue until 15 years after the investment. How will this affect the market equilibrium in traditional rural biotopes, where the costs are the highest? The new equilibrium price is 45,009 €, which is 50% higher compared to the benchmark case of Table 9, while from Table 10 we saw that increasing the trading ratio increases the price only 2%. Furthermore, the land area traded in the market (9365 ha) is 90% less than the benchmark case, whereas with higher trading ratio, the land area increases 55%. The implications of an advanced credit release policy can be found on Appendix A.

Table 11 shows the total estimated size of the offset market for each selected habitat and land areas traded to the year 2040. However, it must be noted that this we did not estimate the entire hypothetical offset market in Finland – only the three selected habitat types.

When the trading ratio is equal to 1 and there is no time delay, the market size is estimated to be almost 370 million € in total. Approximately 37,000 ha of land would be restored and conserved. If the trading ratio increased to 1.6, the total size of the market would be 580 million € and approximately 55,000 ha.

4.2. Market Equilibrium When the Outcomes of Restoration Vary

The results of the expert survey showed that the outcomes of restoration differ between habitats. Now, we take the results into account. To calculate a representative trading ratio, we employ the average ecological value produced with restoration. We set the amount of loss arbitrarily equal to 0.5. Table 12 shows the losses and gains as well as trading ratios in result, both with and without time delay.

Table 12 illustrates clearly that a perfect and well-functioning market with a sufficient stock of mature restored habitats leads to lower trading ratios and expectedly lower market prices. In all, the trading ratios required are the highest in pine mires, because the increase in ecological value achieved with restoration is the lowest in mires. Next, the properties of market equilibrium are examined more closely. Picking up the trading ratios from Table 12 leads to the following market equilibrium in Table 13.

Relative to Table 9, offset prices from pine mires and herb-rich forests increase due to increased trading ratios. In terms of land area, the increase is most dramatic in pine mires, because restoration increases their ecological value only slightly and gradually, so that much larger areas are needed relative to other habitats. Since the trading ratios are the highest in pine mires, their restored land is now almost 2 times higher in comparison to the benchmark case. In herb-rich forests, the total area of compensation sites increases more moderately (40%). Landowners' profits behave accordingly: those of pine mires and herb-rich forests increase, while the profits from rural biotopes decrease as their trading ratios fall below unity. The state of rural biotopes improves such a degree (recall Fig. 2) that, unlike in other habitats, it is more likely that the gains are higher than the losses. The restored land area decreases by approximately 20% in comparison with Table 9 but the impact to the price is very small (0.6%).

Next, we add a time delay to the previous analysis and employ the trading ratios reported in Table 12. The new market equilibrium is presented in Table 14.

Again, as the trading ratios increase, the prices, the profits and the total area of compensation sites increase. Now, the trading ratio in rural biotopes also rises above unity. Due to the high trading ratio (4) in pine mires, the area of the compensation sites now covers almost half of the potential restorable area. The same figure is 2% in herb-rich forests and 14% in rural biotopes.

4.3. Sensitivity Analysis

In the previous section, where we took into account the different outcomes of restoration, the improvement in ecological value was measured at a point in time $\tau = 50$. We next examine how changing the point in time when the improvements are accounted for affects the trading ratios.

Table 15 shows that the later the gains are calculated, the higher they become, because with the passage of time, the difference between a restored habitat and a habitat in a business-as-usual scenario increases. Thus, the trading ratios are higher if the gains are calculated at an earlier point in time. When the trading ratios are higher, the equilibrium prices and profits for landowners as well as the restored land areas increase. Table 16 shows how changing τ to 25 affects the market equilibrium.

We see that relative to Table 13, the trading ratios increase, and the increase is especially high in pine mires. The equilibrium price increases by 10% and the area of compensation sites by 60%. In herb-rich forests, the equilibrium price increases by 6% and the increase in the area of compensation sites is 50%. In rural biotopes, the price increases by 1%, but the increase in the area of compensation sites is 35%. Thus, the determination of τ has a significant impact and must be considered when the results of this study are interpreted.

5. Transaction Costs and the Risk of Failure

It has been suggested that an intermediary would be an institution that would help to alleviate transaction costs for market participants (Coggan et al., 2013a; OECD, 2016). In the previous case of perfect markets, these services were assumed to be cost free, but naturally this is not plausible. In this chapter, we consider how a brokerage fee collected by the intermediary affects the market equilibrium. First, the intermediary collects a payment as a fee for the services it provides to reduce market participants' transaction costs. This is added to the offset price. Second, the intermediary estimates the proportion of failed projects and includes a risk premium in the brokerage services, which again shows up in offset prices.

Fig. 5 qualitatively illustrates the impacts of the additional fees. Pricing the transaction costs and risks affects the market as if it were a tax: if levied on buyers the (after-premium) demand curve shifts downwards, and if levied on suppliers the (after-premium) supply curve shifts upwards. In both cases, the price increases to p^{**} and the fee collected by the intermediary is an amount represented by area $p^{**}ABp_s$. Due to higher prices, the area of compensation sites in total will decrease from Q^* to Q^{**}.

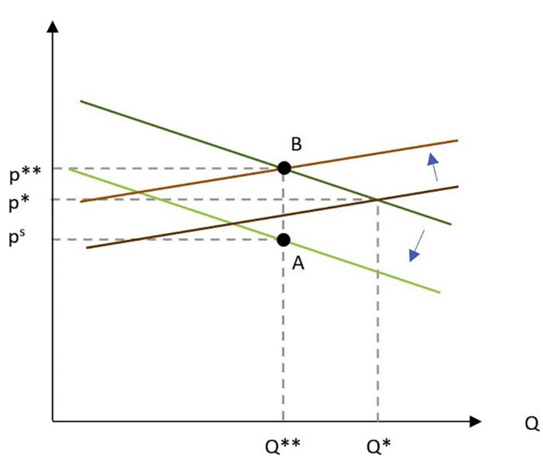

Fig. 5. Effect of additional fees on the market.

5.1. Transaction Costs

In offset markets, transaction costs can incur when the developer must learn about offset requirements, negotiate requirements with the regulator, find suppliers and negotiate contracts with the suppliers. Transaction costs to the supplier can include the costs of learning about offsets and what can be supplied, negotiating contracts with buyers and the regulator, monitoring and reporting compensation measures and responding to enforcement measures in case the compensation sites do not meet their requirements (Coggan et al., 2013a). An intermediary can reduce these costs by providing information, brokerage services etc.

Now, we assume that the intermediary includes an additional payment in the price as a fee for the services it provides to reduce market participants' transaction costs. The fee is 5% of the offset price (from Table 9) (a similar brokerage fee in Hessen, Germany is 6% (OECD, 2016)). The new market equilibrium with the added fee is presented in Table 17.

From the table, we see that the intermediary fee has only a small impact to the market equilibria. Relative to Table 9, the buyer prices increase by 5%. The decrease in the total area of compensation sites is 7–8%. If we compare these figures to those in Table 10, we can see that it is cheaper for the developers to purchase mature offsets from the intermediary and pay the fee, instead of buying immature offsets with a higher trading ratio in pine mires and herb-rich forests. In rural biotopes, where the offset price is higher, a 5% fee causes a 3% higher increase in the price than increasing the trading ratio.

5.2. Risk Premium

Not all restoration projects are likely to succeed. This creates a risk in the market and the environment: buyers buy compensations that do not improve the state of habitats. The intermediary can play a constructive role in reducing the risk of failures in the market. In the previous case of perfect markets, it was implicitly assumed that the intermediary safeguards against failed compensations. The intermediary can price the economic and ecological risks in the brokerage services by estimating the monetary value of the failed projects.

The Monte Carlo simulation results provide data on the percentage of restored sites that will not recover as expected. When restoration is not successful, the sites are not saleable. The intermediary calculates the revenue loss from the failures and allocates a risk premium in the market. We assume that if the outcome of restoration is < 90% of the expected value, restoration has failed and there is no compensation to be sold. For instance, in pine mires, this means that 20% of the restored area, 6279 ha, is useless and the loss is worth approximately 50 million €. The loss per hectare is 1630 €, which is the risk premium collected by the intermediary. In herb-rich forests, 10% of the sites are not saleable. Since the improvement achieved with restoration is so substantial in rural biotopes, we assume that sites that achieve at least 85% of the expected value would be saleable as compensation. Thus, 20% of the sites are not saleable. The new market equilibrium with risk premiums (and trading ratio equal to 1) is presented in Table 18.

Comparing Table 18 with Table 9 reveals that in all habitats, the prices are now 10–20% higher. In pine mires and rural biotopes, the restored land areas are approximately 30% lower. In herb-rich forests, the compensated land area is 13% smaller. Thus, the impact of the elimination of ecological risks is not especially large in the market but by eliminating failures in restoration its impact on biodiversity may be considerable. Economic risk is greatest in rural biotopes (approximately 20 M€), and the risk in terms of land area is greatest in pine mires (6300 ha).

Finally, we consider how increasing the risk premium affects the market equilibrium. Above, we assumed that if the restoration outcome is < 90% of the expected value, or 85% in rural biotopes, restoration has failed and there is no compensation to be sold. Results showing how increasing this requirement by 5% will affect risk premiums and the market equilibrium are provided in Table 19.

In comparison with Table 18, we see that the impact is substantial: the risk premiums increase by 75% in pine mires and 50% rural biotopes and are 2.5 times higher in herb-rich forests. Consequently, the buyer prices increase and the profits decrease further. In pine mires, the profits start to approach zero. If the risk premiums are high, in habitats where there is a lot of uncertainty regarding the success of restoration, offset credits will be expensive and landowners will get negative profits. Thus, it may not be feasible to raise the level of the premiums excessively high.

6. Discussion

Internationally, restoring ecosystems has become an important way to slow down the loss of biodiversity and maintain ecosystem services (Wende et al., 2018). In this article, we developed an equilibrium model to examine biodiversity offset markets and applied the analytical model to three selected habitats. We analysed how trading ratios, the presence of an intermediary and the realization of risks associated with uncertainty affect the market equilibrium: the offset prices and land areas traded as compensation.

The results show that the size of the offset markets could potentially be considerable, which is a prerequisite for a functioning biodiversity offset market (Wissel and Wätzold, 2010). Providing offsets could be a profitable business for landowners as there is potential demand and there would be enough land and suitable habitats for compensations in Finland, even when the trading ratios are relatively high. In habitats where restoration or nature management is laborious and expensive, the offset prices are high and, especially when continuous management is required, compensation can be very costly. The relative amount of biodiversity losses and restored gains as well as possible time delays impact the trading ratios and, thus, have a major impact on the market equilibrium in our analysis, which corresponds to the results found by Doyle and Yates (2010).

An intermediary that provides brokerage, offset aggregator and banker services may decrease the costs of compensation for developers, provided that it can provide mature offsets when needed. The results show that as long as the brokerage fees and risk premiums collected by the intermediary are not excessively high, the impact of pricing these services in the market is quite modest, apart from rural biotopes where the market size may decrease considerably. Drechsler and Hartig (2011) find that long restoration time lags may limit offset credit supply and shrink the size of the market, which also calls for the presence of an intermediary in the market.

The trading ratios applied in our analysis were relatively low in comparison with those found in the literature (Gibbons et al., 2015; Laitila et al., 2014; Moilanen et al., 2009), because expected values were used in the calculation of the trading ratio. We did not consider uncertainty or the possibility that restoration could fail completely – this scenario was included in the risk premium, since the intermediary bears the risk of failure. Secondly, there are also other sources of uncertainty that would increase the trading ratio. They can be taken into account by adding an error weight (Moilanen et al., 2009). Small error weights can be used if there is abundant experience and knowledge regarding the restoration and management of the habitats studied and the site is well surveyed. A higher error is needed if an area is poorly surveyed or there is a lack of knowledge, e.g. if a new restoration technique is tested. The trading ratio increases substantially if it is assumed that success between distinct restoration sites is correlated to some degree. However, the feasibility of very high trading ratios (increasing from dozens to hundreds) is debatable. The trading ratios employed here were consistent with those used in practice (Bull et al., 2016b), except for the fact that the ratios proposed are rarely below 1.0.

There are some limitations in the model and its application. A major challenge was that this type of offset markets had not yet been established in Finland, and the data available on the existing offset markets, realized costs and prices were very limited. To estimate the supply and demand, we had to rely on expert assessments, and apply and combine information from many documented sources. Many assumptions had to be made to include the crucial factors in the study without any support from similar analyses or experiences from existing offset markets.

Determining τ, the point in time when improvement in the ecological state of the habitat is calculated, significantly impacted the trading ratios and, thus, the equilibrium prices and compensation sites traded. This must be noted when the results are interpreted. In the sensitivity analysis, we compared several alternatives (25, 50 and 100 years), and the differences in the equilibrium prices and land areas traded were substantial. However, there is no unambiguous answer to what the level of τ should be. Expert assessments may be the best way to ensure that τ is set to a point in time that is low enough to be feasible, but high enough to ensure that the calculation of ecological gains is reliable. The same applies to the level of intermediary fees and risk premiums. Since there is little data available on the level of these kinds of payments from existing offset markets, we used a similar brokerage fee as in Hessen, Germany. However, the level of the payments is determined in the market. Thus, the results concerning the impact of risk premiums and intermediary fees should only be used to analyse their impact on the market in general.

In our analysis, we assumed that offsetting is mandatory – all adverse impacts on biodiversity from land-use change must be compensated. If offsetting were voluntary, it would strongly affect the demand, and the market size would shrink. We have assumed trading in-kind, but if trading up was possible, purchasing credits from rural biotopes and herb-rich forests could increase, because they are more valuable to biodiversity than mires and thus, provide higher ecological gains. However, high offset prices would likely limit trading up. Trading down is not preferred, but if it was allowed, there would be risk that demand would channel predominantly to pine mires, since they are up to eight times cheaper than other habitats examined in this analysis.

This study aimed to provide a new type of analysis of biodiversity offset markets on the market level. The analytical model introduced here could be used to further study the various factors in the market: taking a closer look at trading ratios or adding carbon offsets in the market, for instance. The model could be further developed and tested with a case study if data on the realized offset trades were to become available. Since the information available on intermediaries in offset markets is limited to a few case studies, closer analysis is needed, e.g. of the various roles of the intermediary and how the intermediaries impact transaction costs and prices and ease trades in the market.

Implementing an offsetting mechanism could improve the current state of biodiversity and habitat restoration in Finland. There is a lot of experience and knowledge regarding the restoration and management of mire and forest habitats in Finland (Aapala et al., 2013; Similä and Junninen, 2011), as well as degraded habitats suitable for restoration, which is an advantage. The results show that there may be potential for both supply of and demand for biodiversity offsets in Finland. Still, offsetting alone will not be sufficient. Preserving the most valuable species and habitats is essential, and all impacts cannot be compensated. Irreplaceable, extremely vulnerable ecosystems and habitats or endangered species are always no-go areas where offsetting cannot be applied. Biodiversity offsetting can potentially be an important addition to the policy mix to halt the alarming rate of biodiversity loss and ensure the well-functioning future ecosystem services.

Declarations of Interest

None.

KONE FOUNDATION

Appendices A and B. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolecon.2019.01.003.

References

Aapala, K., Similä, M., Penttinen, K. (Eds.), 2013. Handbook for the Restoration of Drained Peatlands (in Finnish). Nature Protection Publications of Metsähallitus. Series B 188.

Bamber, J.L., Aspinall, W.P., 2013. An expert judgement assessment of future sea level rise from the ice sheets. Nat. Clim. Chang. 3 (4), 424–427.

Barbour, M.T., 1991. Stream surveys-the importance of the relation between habitat quality and biological condition. In: Sediment and Stream Water Quality in a Changing Environment: Trends and Explanation.

BenDor, T.K., Guo, T., Yates, A.J., 2014. Optimal advanced credit releases in ecosystem service markets. Environ. Manag. 53 (3), 496–509.

Biedermann, R., 2003. Body size and area-incidence relationships: is there a general pattern? Glob. Ecol. Biogeogr. 12 (5), 381–387.

Bonds, M.H., Pompe, J.J., 2003. Calculating wetland mitigation banking credits: adjusting for wetland function and location. Nat. Resour. J. 43, 961.

Bull, J.W., Suttle, K.B., Gordon, A., Singh, N.J., Milner-Gulland, E.J., 2013. Biodiversity offsets in theory and practice. Oryx 47 (03), 369–380.

Bull, J.W., Gordon, A., Watson, J.E., Maron, M., 2016a. Seeking convergence on the key concepts in 'no net loss' policy. J. Appl. Ecol. 53 (6), 1686–1693.

Bull, J.W., Lloyd, S.P., Strange, N., 2016b. Implementation gap between the theory and practice of biodiversity offset multipliers. Conserv. Lett. 10 (6), 656–669.

Business and Biodiversity Offsets Programme (BBOP), 2012. Guidance Notes to the Standard on Biodiversity Offsets.

Calvet, C., Napoléone, C., Salles, J.-M., 2015a. The biodiversity offsetting dilemma: between economic rationales and ecological dynamics. Sustainability 7, 7357–7378.

Calvet, C., Ollivier, G., Napoléone, C., 2015b. Tracking the origins and development of biodiversity offsetting in academic research and its implications for conservation: a review. Biol. Conserv. 192, 492–503.

Coggan, A., Buitelaar, E., Whitten, S.M., Bennett, J., 2013a. Intermediaries in environmental offset markets: actions and incentives. Land Use Policy 32, 145–154.

Coggan, A., Buitelaar, E., Whitten, S.M., Bennett, J., 2013b. Factors that influence transaction costs in development offsets: who bears what and why? Ecol. Econ. 88, 222–231.

Doyle, M.W., Yates, A.J., 2010. Stream ecosystem service markets under no-net-loss regulation. Ecol. Econ. 69 (4), 820–827.

Drechsler, M., Hartig, F., 2011. Conserving biodiversity with tradable permits under changing conservation costs and habitat restoration time lags. Ecol. Econ. 70 (3), 533–541.

Drechsler, M., Wätzold, F., 2009. Applying tradable permits to biodiversity conservation: effects of space-dependent conservation benefits and cost heterogeneity on habitat allocation. Ecol. Econ. 68 (4), 1083–1092.

Fahrig, L., 2001. How much habitat is enough? Biol. Conserv. 100 (1), 65–74.

Fernandez, L., Karp, L., 1998. Restoring wetlands through wetlands mitigation banks. Environ. Resour. Econ. 12 (3), 323–344.

Gamarra, M.J.C., Lassoie, J.P., Milder, J., 2018. Accounting for no net loss: a critical assessment of biodiversity offsetting metrics and methods. J. Environ. Manag. 220, 36–43.

Gardner, T.A., von Hase, A., Brownlie, S., Ekstrom, J.M., Pilgrim, J.D., Savy, C.E., Stephens, T., Treweek, J., Ussher, G.T., Ward, G., ten Kate, K., 2013. Biodiversity offsets and the challenge of achieving no net loss. Conserv. Biol. 27 (6), 1254–1264.

Gibbons, P., Evans, M.E., Maron, M., Gordon, A., Le Roux, D., von Hase, A., Lindenmayer, D.B., Possingham, H.P., 2015. A loss-gain calculator for biodiversity offsets and the circumstances in which no net loss is feasible. Conserv. Lett. 9 (4), 252–259.

Gibbons, P., Macintosh, A., Constable, A.L., Hayashi, K., 2018. Outcomes from 10 years of biodiversity offsetting. Glob. Chang. Biol. 24 (2), e643–e654.

Hartig, F., Drechsler, M., 2009. Smart spatial incentives for market-based conservation. Biol. Conserv. 142, 779–788.

Kansallinen energia- ja ilmastostrategia, 2013. Valtioneuvoston selonteko eduskunnalle 20. päivänä maaliskuuta 2013 VNS 2/2013 vp. Työ- ja elinkeinoministeriön julkaisuja. Energia ja ilmasto 8/2013.

Kemppainen, L., Lehtomaa, L., 2009. Perinnebiotooppien hoidon tila ja tavoitteet. Valtakunnallinen kooste perinnebiotooppien alueellisista hoito-ohjelmista. Lounais-Suomen ympäristökeskuksen raportteja 2/2009.

Kolstad, C.D., 2000. Environmental Economics. Oxford University Press, New York.

Kotiaho, J.S., Kuusela, S., Nieminen, E., Päivinen, J., 2015. Elinympäristöjen tilan edistäminen Suomessa. Suomen ympäristö. 8/2015. (A shorter version available in English: Kotiaho, J.S., Kuusela, S., Nieminen, E., Päivinen J. & Moilanen, A. (2016). Framework for assessing and reversing ecosystem degradation – Report of the Finnish restoration prioritization working group on the options and costs of meeting the Aichi biodiversity target of restoring at least 15 percent of degraded ecosystems in Finland. Reports of The Ministry of The Environment 15en | 2016).

Laitila, J., Moilanen, A., Pouzols, F.M., 2014. A method for calculating minimum biodiversity offset multipliers accounting for time discounting, additionality and permanence. Methods Ecol. Evol. 5 (11), 1247–1254.

Leinonen, A. (Ed.), 2010. Turpeen tuotanto ja käyttö. Yhteenveto selvityksistä, (VTT Tiedotteita – Research Notes 2550. 104 s. (toim.)).

Levrel, H., Scemama, P., Vaissière, A.C., 2017. Should we be wary of mitigation banking? Evidence regarding the risks associated with this wetland offset arrangement in Florida. Ecol. Econ. 135, 136–149.

Maron, M., Hobbs, R.J., Moilanen, A., Matthews, J.W., Christie, K., Gardner, T.A., Keith, D.A., Lindenmayer, C.A., McAlpine, C.A., 2012. Faustian bargains? Restoration realities in the context of biodiversity offset policies. Biol. Conserv. 155, 141–148.

Maron, M., Ives, C.D., Kujala, H., Bull, J.W., Maseyk, F.J., Bekessy, S., Gordon, A., Watson, J.E.M., Lentini, P.E., Gibbons, P., Possingham, H.P., Hobbs, R.J., Keith, D.A., Wintle, B.A., Evans, M.C., 2016. Taming a wicked problem: resolving controversies in biodiversity offsetting. Bioscience 66 (6), 489–498.

McKenney, B.A., Kiesecker, J.M., 2010. Policy development for biodiversity offsets: a review of offset frameworks. Environ. Manag. 45 (1), 165–176.

Millennium Ecosystem Assessment (MEA), 2005. Ecosystems and Human Well-being: Synthesis. Island Press, Washington DC (137 pages).

Moilanen, A., Van Teeffelen, A.J., Ben-Haim, Y., Ferrier, S., 2009. How much compensation is enough? A framework for incorporating uncertainty and time discounting when calculating offset ratios for impacted habitat. Restor. Ecol. 17 (4), 470–478.

Natural Resources Institute Finland (LUKE), 2016. Volumes and Prices in Roundwood Trade, 9/ 2016. (27.10.2016).

Organization for Economic Cooperation and Development OECD, 2016. Biodiversity Offsets: Effective Design and Implementation. OECD Publishing, Paris.

Quétier, F., Lavorel, S., 2011. Assessing ecological equivalence in biodiversity offset schemes: key issues and solutions. Biol. Conserv. 144 (12), 2991–2999.

Rassi, P., Hyvärinen, E., Juslén, A., Mannerkoski, I. (Eds.), 2010. Suomen lajien uhanalaisuus – Punainen kirja 2010. Ympäristöministeriö & Suomen ympäristökeskus, Helsinki (685 s. (toim.)).

Raunio, A., Schulman, A., Kontula, T. (Eds.), 2008. Suomen luontotyyppien uhanalaisuus – Osa 2: Luontotyyppien kuvaukset. Suomen ympäristökeskus, Helsinki (Suomen ympäristö 8/2008. 572 s (toim.)).

Similä, M., Junninen, K. (Eds.), 2011. Metsien ennallistamisen ja luonnonhoidon opas. Metsähallituksen luonnonsuojelujulkaisuja. Sarja B 157, (toim.).

Tiitu, M., Helminen, V., Järvenpää, E., Härmä, P., Hatunen, S., Rehunen, A., 2015. Rakennetun alueen pinta-alan ennakointi. Paikkatietoaineistojen ja -menetelmien hyödyntäminen rakennetun alueen muutosten laskennassa. Suomen ympäristökeskuksen raportteja 28/2015.

Tjørve, E., 2003. Shapes and functions of species–area curves: a review of possible models. J. Biogeogr. 30 (6), 827–835.

Vaissière, A.C., Levrel, H., 2015. Biodiversity offset markets: what are they really? An empirical approach to wetland mitigation banking. Ecol. Econ. 110, 81–88.

van Teeffelen, A.J., Opdam, P., Wätzold, F., Hartig, F., Johst, K., Drechsler, M., Vos, C.C., Wissel, S., Quétier, F., 2014. Ecological and economic conditions and associated institutional challenges for conservation banking in dynamic landscapes. Landsc. Urban Plan. 130, 64–72.

Wende, W., Tucker, G., Quétier, F., Rayment, M., Darbi, M. (Eds.), 2018. Introduction: Biodiversity Offsets – The European Perspective on No Net Loss of Biodiversity and Ecosystem Services. Springer, Cham.

Wissel, S., Wätzold, F., 2010. A conceptual analysis of the application of tradable permits to biodiversity conservation. Conserv. Biol. 24 (2), 404–411.