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Central European Journal of Economic Modelling and Econometrics

Ecosystem Services in Competing Land Use Model with Infrastructure Effect

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Submitted: 19.01.2019, Accepted: 21.06.2019

Abstract

The development of linear infrastructure increases the degree of fragmentation of natural areas and has a negative impact on biodiversity and the range of available ecosystem services. The basic competing land use model is expanded to include infrastructure development. The extended model leads to the conclusion that due to the dual impact of the infrastructure (lowering the value of ecosystem services and increasing the private rents to developed land), the size of the natural area in the long-term equilibrium will be lower compared to the basic model. The preservation of nature ceases to be profitable enough. Infrastructure also reduces the marginal costs of conversion and thus increasing the volume of natural land being converted at avery moment along the transition path. If the decisions on optimal management of natural areas and infrastructure development are undertaken together, the result is a lower density of the infrastructure network and a larger ecosystem area in the steady state.

Keywords: sustainability, land use change, optimal land conversion, ecosystem services

JEL Classification: C61, Q24, Q57

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1 Introduction

In a purely economic perspective, the conversion of natural areas in the process of economic development can be seen as a change of one form of assets to another offering greater economic benefits – see (Parks *et al.*, 1998) for example of static marginal analysis of land allocation between forest and non-forest use. Competing land use model was used to describe the deforestation process and the subsequent increase in forest cover during the course of long-run economic development - see (Hartwick *et al.*, 2001; Barbier *et al.*, 2010; Barbier, 2011; Barbier and Tesfaw, 2015) and (Barbier *et al.*, 2017). The concept of forest transition was first described by Mather (1992) in the early 90s and later was tested empirically by numerous researchers (Wolfensberger *et al.*, 2015).

The main purpose of this paper is the theoretical consideration of the impact of linear infrastructure on the process of conversion of natural areas within the framework of basic competing land use model. To my best knowledge, the impact of linear infrastructure has not been explicitly recognized as a key factor determining the optimal development of natural ecosystems. The paper indicates the necessity of integrated spatial management. If the decisions on optimal management of natural areas and infrastructure development are undertaken together, the result is a lower density of the infrastructure network and a larger ecosystem area in the steady state. Another important conclusion is the need to consider an infrastructure factor when implementing and monitoring payments for ecosystem services programs.

The paper is essentially a theoretical sketch. The basic model as in (Hartwick *et al.*, 2001; Barbier, 2011) was used as a starting point for further theoretical considerations. Numerical simulations are possible (and needed), but it requires expert knowledge to define appropriate functional forms and parameters. The rest of the paper is organized as follows. Section 2 briefly discusses the main concepts and provides basic theoretical and empirical background for modelling developed in futher sections. Section 3 describes the basic competing land use model as proposed by Hartwick *et al.* (2001) and Barbier (2011) and consider payments for ecosystem services (PES) within the discussed framework. In section 4 the basic competing land us model is expanded to include infrastructure development. The ending contains the main conclusions and suggestions for further research in this area.

2 Theoretical background

Natural capital is a relatively new concept, developed since the early 1990s. Pearce and Turner (1990) used the term natural capital without giving its detailed definition. It is understood very broadly as natural resources, including tropical forests, ocean habitats, wetlands, fishery, atmosphere and stratosphere, etc. According to R. Costanza and H. Daly, natural capital is an extension of the economic concept of capital as *a stock that yields a flow of valuable goods or services into the future* for environmental goods and services (Costanza and Daly, 1992). Natural capital in the form of ecosystems yields numerous and diverse ecosystem services for both production and consumption and, above all, for the maintenance of life on Earth (Costanza *et al.*, 1997). In addition to natural resources, ecosystems provide, to name just a few, life-support, regulatory, cultural and aesthetic benefits (Daily, 1997). Interdisciplinary research points to the key role of biodiversity in preserving the capacity of ecosystems to generate benefits for people. Loss of biodiversity threatens the sustainability of ecosystems and can have a significant impact on the ability of the ecosystem to provide the services (Diaz, 2006; Cardinale, 2012). Research shows that indicators for biodiversity and aggregated ecosystem services supply are positively related but this relationship is influenced by the spatial trade-offs among provisioning and regulating ecosystem services. However, habitats in a favourable conservation status provide more biodiversity and have higher potential to supply regulating and cultural ecosystem services in particular (Maes *et al.*, 2012).

The Millennium Ecosystem Assessment (2005) distinguishes 31 ecosystem services assigned to four categories: provisioning, regulating, cultural and supporting. As noted over the past 50 years, 15 out of 24 assessed ecosystem services have been degraded or used in ways that threaten sustainability, for instance drinking water supply, fish populations, air and water purification, climate and natural hazards regulation. Taking into consideration the above mentioned facts, the fundamental problem is the assessment and control of changes occurring in natural capital resources and the availability of ecosystem services.

The preservation of remaining resources of natural capital in the context of growing ecological scarcity is largely conditioned by the inclusion of natural capital in the economic account, similarly to other forms of capital. As observed, the main cause of the destruction and degradation of natural ecosystems is the lack of consideration of the value of ecosystem services in decision-making and development policy formation (The Millennium Ecosystem Assessment, 2005). The inclusion of natural capital and ecosystem services into decision-making process is essential for achieving the sustainability goals in the 21st century (Guerry *et al.*, 2015).

The history of introducing the ES concept into the theory of economics, as well as the development of payments for ecosystem services schemes is discussed by Gómez-Baggethun *et al.* (2010). Authors conclude, that the focus on monetary valuation and market-based policy design has contributed much to mainstream ecosystem services science and attract political support for conservation. DeGroot *et al.* (2012) conducted an overview of the existing valuation results (over 320 publications) for the 10 major biomes: open ocean, coral reefs, coastal systems, coastal wetlands, inland wetlands, lakes, tropical forests, temperate forests, woodlands and grasslands. Using data provided by DeGroot *et al.* (2012) and the value of agricultural rents, the value of the total net change in land cover in Poland for the period 2006-2012 was estimated (Telega, 2017). Net unit value was calculated as the difference between the values in 2006 and 2012 and then multiplied by the size of the changed area. For the values of ecosystem services vary considerably, the calculations were made separately for median, minimum and maximum values of ecosystem services. In the years 2006-2012 changes in land cover reached 266391 hectares of Polish land. The share of areas for which values are inaccessible (mainly urban areas) accounts for about 6.7% of the total changed area in 2006-2012. The obtained results (using GIS software) indicate the increase of natural capital resources in Poland in the period 2006-2012, with the value around 1.6 bln int. dollars (2007 price levels).

As noted in The Millennium Ecosystem Assessment (2005), "economic and financial interventions provide powerful instruments to regulate the use of ecosystem goods and services". Payments for ecosystem services (PES) have been listed alongside taxes or user fees, cap-and-trade systems and mechanisms to enable consumer preferences to be expressed through markets as an example of effective tools to stop the degradation of ecosystems. A seminal definition of PES is given by Wunder (2005). PES are characterized by five criteria: "(1) a voluntary transaction where (2) a well-defined service (or a land-use likely to secure that service) (3) is being 'bought' by a (minimum one) ES buyer (4) from a (minimum one) ES provider (5) if and only if the ES provider secures ES provision (conditionality)". In his later work Wunder (2005) corrected the definition, emphasizing the role of conditionality as the single most important PES feature. In this case, PES are defined as "(1) voluntary transactions (2) between service users (3) and service providers (4) that are conditional on agreed rules of natural resource management (5) for generating offsite services".

This approach is referred to as Coasian conceptualization of PES (Engel *et al.*, 2013; Schomers and Matzdorf, 2013). It is noted that the most common payer (buyer) for the services of ecosystems, being a public good, is the government, so it is more appropriate to conceptualize PES in a Pigouvian manner. As Vatn (2010) points out "As low transaction costs are illusory for most environmental goods, the Coasean model is relevant only for a very small number of cases". Schomers and Matzdorf (2013) discuss PES case studies reflecting the Coasean as well as Pigouvian conceptualization, but most examples are in fact governmental payment programs, which can be seen as some form of Pigouvian subsidy. Muradian *et al.* (2010) define PES as "A transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources". This much broader definition, compared to the one proposed by Wunder, underlines the role of the social interest and is definitely Pigouvian in its spirit.

Regardless of the discrepancies in the adopted definitions of PES, it can be considered after Wunder (2005) that "the core idea of PES is that external ES beneficiaries make direct, contractual and conditional payments to local landholders and users in return for adopting practices that secure ecosystem conservation and restoration". In the further part of the work, we will understand PES as an additional payment for the landowner dependent on the size of natural areas, the purpose of which is to create an additional incentive to preserve nature. An optimal incentive policy will result in land

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being put to its "highest and best use", ie. the land use that maximizes total benefits to society, including the value of eco-system services (Polasky *et al.*, 2014). Most prominent examples of PES include programs implemented in Costa Rica (Sanchez-Azofeifa *et al.*, 2007; Pagiola, 2008), Mexico (Munoz-Pina *et al.*, 2008), European and US agri-environmental programs (Schomers and Matzdorf, 2013) as well as eco-compensation regulations in China (e.g. Sloping Land Conversion Program - SLPC) (Li *et al.*, 2008).

Spatial planning and land use decisions are of particular importance for the preservation of ecosystems, yet the majority of land-use decisions often ignore the value of ecosystem services. Bateman (2013) show that significant value increases can be obtained from targeted planning by incorporating all potential services and their values. It is worth noting that it often comes to tradeoffs between different types of ecosystem services. For example, actions to enhance the supply of provisioning services such as food and timber, have led to declines in regulating and cultural services such as nutrient cycling, flood regulation, and opportunities for recreation. Crop and pork production, were found to have the highest number of significant negative correlations with other services (Raudsepp-Hearnea *et al.*, 2010). Goldstein *et al.* (2012) considered various scenarios for land development with positive financial return and pointed out tradeoffs between carbon storage and water quality as well as between environmental improvement and financial return.

Land-change science seeks to understand the human and environment dynamics resulting in changes in land uses and covers in terms of their type, magnitude as well as location (Rindfuss, 2004). Ecosystems are basically a spatial concept, so the natural approach is to use GIS to evaluate land cover changes. Global land use change can mostly be characterized by the expansion of urban and infrastructure areas at the expense of agricultural land and by the expansion of agricultural land at the expense of grasslands, savannahs and forests (UNEP, 2014). In Europe, changes in land use are recorded by European Environment Agency (EEA) using spatial data, e.g. Corine Land Cover – CLC data (EEA, 2006, 2017). Since the coverage of the site may be altered or degraded but also reconstructed, this process is very similar to the transformation of capital resources in the economy. It can therefore be described in terms of flows between different types of land cover. Between 2000 and 2006 a growth in the share of artificial areas was observed, mainly stemming from the demand for housing, services and recreation. In recent years, the dominant trend has been expansion of urban and artificial areas causing loss of natural areas and further fragmentation of existing landscape structures. By now, nearly a third of Europe's landscape is characterized by high fragmentation, which affects the state of ecosystems, their ability to provide services, and the provision of safe habitats for species (EEA, 2015). Fragmentation of natural areas is caused mainly by the development of transport infrastructure and the urban sprawl (EEA, 2011). It is estimated that the extent of the fragmentation of natural areas in Europe is very diverse, but the most fragmented are the Benelux countries, Germany, France, and

Central Europe.
Linear infrastructure (e.g. roads, railways, utility easements and other industrial linear corridors) is of great importance for the economy and has a significant impact on the natural environment. These impacts include habitat loss and degradation, barrier or filter to movement, noise, light, and chemical pollution and disturbance from vehicles, wildlife mortality, attraction of invasive species etc. (van der Ree *et al.*, 2015). The density of the road network determines the scale and intensity of road impacts on the environment. Wildlife populations have long response times (up to several decades) to increases in landscape fragmentation. As the fragmentation progresses, the population does not decrease linearly, but usually exhibits a threshold after which there is a dramatic decline (Jaeger, 2015). Roads also act as drivers of deforestation facilitating access to previously inaccessible forest areas, as well as increasing profitability of agriculture and ranching (Fearnside, 2015). For example, in Amazonia deforestation spreads outwards from highways and their associated access roads.

3 Basic model and policy implications

3.1 Basic model

The basic model is as follows. Consider a certain area divided between natural ecosystems N_0 and developed areas D_0, with the assumption that $N_0 > D_0$. We do not specify here what these developments are, it could be agriculture of housing, depending on context and particular case. Let us assume that the changes are continuous – i.e. the natural area is subject to continuous anthropogenic conversion processes, e.g. farming, infrastructural, housing processes, etc. Changes in land use are irreversible. Formally, this assumption is not necessary. We can consider the analogous framework in which the ecosystem is reproduced, i.e. the reverse conversion takes place. However, irreversibility assumption seems to make sense for other reasons, at least in short to medium terms. In most cases restoration of the natural ecosystem is unlikely due to high costs, as well as technical, legal (e.g. property rights) and natural constraints. After all, planting trees is not fully equivalent to the restoration of the destroyed ecosystem. Denote $c(t)$ the area of the ecosystem being converted at time t. Then

$$N(t) = N_0 - \int_0^t c(s)\,ds, \qquad \dot{N} = -c, \tag{1}$$

$$D(t) = D_0 + \int_0^t c(s)\,ds, \qquad \dot{D} = c. \tag{2}$$

Let $E(N(t))$ and $R(D(t))$ denote respectively the value of ecosystem services and the periodic rent to developed land. Both E and R are concave. Conversion costs (e.g. costs of clearing c hectares of natural forest) are $C(c(t))$. Obviously, as $D(t) = N_0 + D_0 - N(t)$, the value of rent could be expressed as $R(N)$ and $R'(D) = -R'(N)$. We have to make following assumptions to ensure the existence of internal steady state $N^* \in (0, N_0)$ (Hartwick *et al.*, 2001)):

$$C(0) = C'(0) = 0, \quad C'(c) \geq 0, C''(c) > 0. \tag{3}$$

$$E'(0) > R'(D_0 + N_0), \quad E'(N_0) < R'(D_0). \tag{4}$$

As noted by Hartwick *et al.* (2001), the proces of land conversion can be viewed as "sticky stock adjustment" - initial large amount of natural area N_0 is a "disequilibrium". The problem can be simplified to maximizing the net present value of a given area (social welfare function):

$$V = \int_0^{\infty} \left[E(N(t)) + R(N(t)) - C(c(t))\right] e^{-\rho t}\, dt. \tag{5}$$

with the constraint given by (3). We assume that all variables are continuous functions of time t. In order to shorten the notation, the argument t will be skipped. By default, we denote by $\dot{X}$ a derivative of X with respect to t. The current value Hamiltonian for the problem is:

$$H = E(N) + R(N) - C(c) - \lambda c, \tag{6}$$

where λ is a shadow price of natural area N – see Kamien and Schwartz (2012). N is a state variable and c is control. Applying the Pontryagin maximum principle, we obtain the necessary conditions for maximization of social welfare function (5):

$$\frac{\partial H}{\partial c} = 0 \iff C'(c) = -\lambda, \tag{7}$$

$$\frac{\partial H}{\partial N} = \rho\lambda - \dot{\lambda} \iff E'(N) - R'(D) = \rho\lambda - \dot{\lambda}, \tag{8}$$

and standard transversality conditions.
We get the steady state N^* of the model by setting $\dot{N} = 0$. Then $c = 0$ and $C(c) = C'(c) = 0$, therefore $\lambda = 0$ and $\dot{\lambda} = 0$. From (7) and (8) we have

$$\rho C'(c) = R'(D) - E'(N). \tag{9}$$

and

$$\frac{R'(D)}{\rho} = \frac{E'(N)}{\rho}. \tag{10}$$

Considering 4, the steady state value $N^* \in (0, N_0)$ exists. In equilibrium, the marginal benefit from the natural area N is equal to the marginal benefit of development. We can see $\frac{R'(D)}{\rho}$ and $\frac{E'(N)}{\rho}$ as capitalized value or "price" of marginal unit of developed and natural land respectively. Let us note, that if ecosystem benefits are not taken into account, that means $E(N) = E'(N) = 0$, the equilibrium value of N is determined by

$$\frac{R'(D) + \dot{\lambda}}{\rho} = C'(c).$$

As long as $R'(D)$ is positive with increasing D, the whole area $N_0 + D_0$ will be converted.
The stationary point $(N, c) = (N^*, 0)$ is a saddle and for every N_0, there is c_0 that the trajectory $(N(t), c(t))$ is a stable branch of the saddle path. Consider the system of two ODE's

$$\begin{cases} \dot{N} = -c \\ \dot{c} = \dfrac{1}{C''(c)} \left[\rho C'(c) + E'(N) + R'(N)\right] \end{cases} \tag{11}$$

where the second equation is derived from (7) and (8) and $C''(c) \neq 0$. Then the stationary point is $(N^*, 0)$, where $E'(N^*) = -R'(N^*)$. The Jacobian for the system is

$$J = \begin{bmatrix} 0 & -1 \\ \dfrac{E''(N) + R''(N)}{C''(c)} & -\dfrac{C'''(c)}{(C''(c))^2}\left[\rho C'(c) + E'(N) + R'(N)\right] + \rho \end{bmatrix}. \tag{12}$$

As E and R are concave, $|J| = \dfrac{E''(N) + R''(N)}{C''(c)} < 0$ and both eigenvalues

$$\lambda_{1,2} = \frac{1}{2}\left(\mathrm{Tr}J \pm \sqrt{(\mathrm{Tr}J)^2 - 4|J|}\right) \tag{13}$$

are real numbers with opposite signs that indicates a saddle-path stability. Appendix 1 presents a simple numerical simulation assuming that the benefit functions are square root, while the cost function is quadratic. Stability analysis was performed as well as elements of comparative statics for a stationary point.

3.2 Model with benefits from land clearing

Hartwick *et al.* (2001) considered quite similar problem with additional benefits coming from the timber taken from the cleared land. If P is a price of timber (or any other product obtained in the conversion process) we have an additional benefits Pc to be included in the optimization framework. Hamiltonian function takes form $H = E(N) + R(N) + Pc - C(c) - \lambda c$ and necessary condition (7) becomes:

$$P - C'(c) = \lambda. \tag{14}$$

The steady state value N^{**} is given by the equation

$$C'(c) = P + \frac{R'(D)}{\rho} - \frac{E'(N)}{\rho} = 0 \iff P + \frac{R'(D)}{\rho} = \frac{E'(N)}{\rho}. \tag{15}$$

To ensure the existance of steady state value N^{**} we must assume that $E'(0) > R'(D_0 + N_0) + \rho P, \quad E'(N_0) < R'(D_0) + \rho P$. Note, that $N^{**} < N^*$. Additional benefits of clearing result in a smaller area of ecosystems in equilibrium.

3.3 Optimal policy

Rational policy of ecosystem preservation should be also based on economic criteria. Obviously, the valuation of ecosystem services is still a major problem (TEEB, 2010). In spite of these difficulties, the basic competing land use model allows us to formulate criteria for economic rationality in the management of natural capital. With simplifying assumption that the annual value of ecosystem services provided by particular biome is not time and space dependent, ie. the value is simply proportional to its area, the discussed model undergoes considerable simplification and deciding how to use the given area comes to comparing the two NPV alternatives (Telega, 2017). The conversion of the ecosystem is economically justified when the shadow price of the ecosystem is negative, i.e., capitalized rent value to developed land (for agricultural, residential and other purposes) exceeds the capitalized value of the unit of ecosystem services. Conversely, if the shadow price of the ecosystem is positive, it is advisable to restore the areas previously developed to their natural state (if possible). As has been noted, land development will not be optimal if the value of environmental services is not taken into account. Because in most cases ecosystem services are public goods, this result should be considered the most likely. PES should therefore be an instrument that "enforces" the optimal result.
Let p denote the payment per unit of natural area. Additionally, let's introduce a distinction between private $E_p(N)$ and social $E_s(N)$ ecosystem benefits, which are external to the decision maker. Let $E(N) = E_p(N) + E_s(N)$ for simplisity. Then the goal of the landowner is to maximize

$$V = \int_0^\infty \left[E_p(N(t)) + R(N(t)) - C(c(t)) + pN\right] e^{-\rho t}\, dt, \tag{16}$$

where p is exogenous to the landowner, ie. p is a constant value determined by regulatory authority. Then the steady state value N^* is given by the slightly modified equation 10:

$$\frac{R'(D^*)}{\rho} = \frac{E_p'(N^*) + p}{\rho}. \tag{17}$$

The optimal outcome could be reached by setting $p = p_1 = E_s'(N^*) = E'(N^*) + -E_p'(N^*)$, which is very Pigouvian in its nature. If the value of ecosystem services for landowner is zero, then simply $p_1 = E_s'(N^*) = E'(N^*)$. The dynamics of the model is analogous to the one discussed in section 3.1, ie. saddle-path stability.

4 Extended model with infrastructure development

4.1 Exogenous infrastructure development

In this work, we will consider a linear infrastructure (e.g. roads, railways, utility easements and other industrial linear corridors) that occupies a relatively small area. We will assume that the area of land occupied by the infrastructure is negligible. However, it is of great importance for the economy and has a significant impact on the natural environment as mentioned in section 2.

Let's consider the role of infrastructure within the framework of competing land use model. The impact of infrastructure development on the above-mentioned optimal conversion process can be twofold. On the one hand, the development of linear infrastructure (roads, transport routes etc.) increases the degree of fragmentation of natural areas. As noted above, this usually has a negative impact on biodiversity and the range of available ecosystem services. On the other hand, the infrastructure provides measurable social benefits that are largely external to the area under consideration, as well as private benefits in the form of higher rents. Finaly, the infrastructure reduces the operating costs in a given area, which can lead to a reduction in the cost of conversion C for each c.

We can now modify the model discussed in the previous chapter to include the role of the infrastructure. Let $I \in [1, I^{max}]$ denote the size of the infrastructure in a given area. We can see I as a density of linear infrastructure.

First, let us consider a situation in which decisions on the expansion of infrastructure and decisions on the management of natural areas are taken separately. This is a situation close to the observed reality - we can assume that infrastructural decisions are made at a higher decision level. Let $Q(I(t))$, $Q'(I) > 0$, $Q''(I) < 0$, be the net benefit of the infrastructure (benefits minus maintenance costs). Let $K(k(t))$ denote the infrastructure construction costs and $\dot{I} = k$. We assume that K is increasing and strictly convex and $K(0) = K'(0) = 0$. We can consider a simple optimization model, where the goal is to determine the optimal size of infrastructure. We have

$$W = \int_0^\infty [Q(I(t)) - K(k(t))]e^{-\rho t}\, dt, \tag{18}$$

I is a state variable and k is control. The current value Hamiltonian for the problem is

$$H = Q(I) - K(k) + \mu k. \tag{19}$$

The necessary conditions are:

$$\frac{\partial H}{\partial k} = 0 \iff K'(k) = \mu, \tag{20}$$

$$\frac{\partial H}{\partial I} = \rho\mu - \dot{\mu} \iff Q'(I) = \rho\mu - \dot{\mu}. \tag{21}$$

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Combining both we get $\dfrac{Q'(I)+\dot{\mu}}{\rho} = K'(k)$. At the steady state $\dot{I} = k = 0$ and $\mu = \dot{\mu} = 0$, so $Q'(I) = 0$. As long as net marginal benefits of infrastructure are positive I increases up to level I^{max} for which $Q'(I^{max}) = 0$.
The main variables of the model, i.e. the benefits of ecosystems E, rent to developed land R and conversion costs C, become functions of two variables. The flow of ecosystem benefits E is a function of the total natural area N and infrastructure density I being the proxy for the degree of fragmentation. Formally, we can assume that

$$\tilde{E} = \tilde{E}(N, I), \quad \frac{\partial \tilde{E}}{\partial N} > 0, \quad \frac{\partial \tilde{E}}{\partial I} < 0. \tag{22}$$

Similarly we have

$$\tilde{R} = \tilde{R}(N, I), \quad \tilde{R}(N, 0) = 0, \quad \frac{\partial \tilde{R}}{\partial N} < 0, \quad \frac{\partial \tilde{R}}{\partial I} > 0, \tag{23}$$

and

$$\tilde{C} = \tilde{C}(c, I), \tilde{C}(0, I) = 0, \ \frac{\partial \tilde{C}}{\partial c} > 0, \quad \frac{\partial \tilde{C}}{\partial c}(0, I) = 0, \quad \frac{\partial \tilde{C}}{\partial I} \leq 0. \tag{24}$$

$I = I^{max}$ is a constant, i.e. I is a parameter describing the state of the linear infrastructure in a given area. We define functions $\tilde{E}, \tilde{R}, \tilde{C}$ as follows:

$$\tilde{E} = \frac{E(N)}{I^{\alpha}}, \quad \tilde{R} = R(D)I^{\beta}, \quad \tilde{C} = \frac{C(c)}{I^{\gamma}}, \tag{25}$$

where $\alpha, \beta, \gamma > 0$. All previous assumptions about the functions E, R, C remain valid. The adoption of such functional forms results from the mathematical convenience and practice of using a multiplicative function (eg Cobb-Douglas function) in economic modeling. In this case, we can "produce" the economic good R with two production factors: land and infrastructure. When $I = 1$, i.e. infrastructure density is at its minimum, we have the model described in the previous chapter. Necessary conditions for optimization now are:

$$\frac{\partial H}{\partial c} = 0 \iff \frac{C'(c)}{I^{\gamma}} = -\lambda, \tag{26}$$

$$\frac{\partial H}{\partial N} = \rho\lambda - \dot{\lambda} \iff \frac{E'(N)}{I^{\alpha}} - R(D)I^{\beta} = \rho\lambda - \dot{\lambda}. \tag{27}$$

Combining (26) and (27) we get

$$\frac{E'(N)}{\rho} = \frac{R'(D)}{\rho} I^{\beta+\alpha}. \tag{28}$$

Starting from D_0, the developed area $D(t)$ grows until $R'(D)I^{\beta+\alpha}$ decreases to $E'(N)$. Because $I^{\beta+\alpha} > 1$, the steady state value $\tilde{N} < N^*$. The stronger the impact of infrastructure (parameters α and β) on the volume of ecosystem services and economic rents, the smaller the size of natural areas N in equilibrium. Note that there is a double effect here - the infrastructure increases the scope of economic benefits and reduces the size of ecosystem services. Thus, the preservation of nature ceases to be "profitable". Infrastructure also reduces the marginal costs of conversion, thus the converted area $c(t)$ is higher for every t.

4.2 Endogenous infrastructure development

Now, let's consider the process of infrastructure development and ecosystem conversion in one framework. We have

$$J = \int_0^\infty \left[\tilde{E}(N(t), I(t)) + \tilde{R}(N(t), I(t)) - \tilde{C}(c(t), I(t)) + Q(I(t)) - K(k(t))\right] e^{-\rho t}\, dt \tag{29}$$

N and I are state variables, c and k are control. Substituting (25), the current value Hamiltonian for the problem is

$$H = E(N)I^{-\alpha} + R(N)I^{\beta} - C(c)I^{-\gamma} + Q(I) - K(k) - \lambda c + \mu k.$$

Necessary conditions for optimization now are:

$$\frac{\partial H}{\partial c} = 0 \iff \frac{C'(c)}{I^{\gamma}} = -\lambda, \quad \frac{\partial H}{\partial k} = 0 \iff K'(k) = \mu, \tag{30}$$

$$\frac{\partial H}{\partial N} = \rho\lambda - \dot{\lambda} \iff \frac{E'(N)}{I^{\alpha}} - R'(D)I^{\beta} = \rho\lambda - \dot{\lambda} \tag{31}$$

$$\frac{\partial H}{\partial I} = \rho\mu - \dot{\mu} \iff Q'(I) - \alpha E(N)^{-\alpha-1} + \beta R(D)I^{\beta-1} + \gamma C(c)I^{-\gamma-1} = \rho\mu - \dot{\mu}. \tag{32}$$

At the steady state $\dot{N} = c = 0$ and $\dot{I} = k = 0$. Then $\lambda = \mu = 0$ and

$$\begin{cases} \frac{E'(N^*)}{(I^*)^{\alpha}} = R'(D^*)(I^*)^{\beta} \\ Q'(I^*) + \beta R(D^*)I^{\beta-1} = \frac{\alpha E(N^*)}{(I^*)^{\alpha+1}} \end{cases} \tag{33}$$

If $\dfrac{\alpha E(N^*)}{(I^*)^{\alpha+1}} - \beta R(D^*)I^{\beta-1} > 0$ (ie. marginal ecosystem services loss from infrastructure development is greater than marginal private benefits gain), then we have $I^* < I^{max}$ and $\tilde{N} < N^*$ in equillibrium. If the decisions on optimal management of natural areas and infrastructure development are undertaken together, the result is a lower density of the infrastructure network and a larger ecosystem area. Of course, ecosystem services are the essence of the problem. Lost ecosystem services due to increased fragmentation reduce the net benefits of infrastructure, thus the optimal density of infrastructure is lower than I^{max}. Obviously, if the ecosystem services are not taken into account at all, then $N = 0$ and $I = I^{max}$ in equilibrium. Unfortunately, this result can in many cases be considered a typical scenario, although it may not always be so extreme. If $\frac{\alpha E(N^*)}{(I^*)^{\alpha+1}} - \beta R(D^*)I^{\beta-1} < 0$ it is possible to obtain $N > 0$ and $I > I^{max}$ in the steady state. We reject this option by assumption.

4.3 Optimal policy

Let's consider PES within the extended framework with infrastructure. We can consider two situations. The first, when the landowner is the decision-maker (infrastructural decisions are not taken into account) and the government is the payer for ecosystem services. The second situation is when the government is the decision-maker, undertaking joint decisions on the development of natural areas and infrastructure development, while the payer for ecosystem services is another state or group of states.
In the first case, let again $E(N) = E_p(N)+E_s(N)$. The relevant equation determining the steady state $\tilde{N}$ takes the form

$$\frac{E_p'(\tilde{N})}{I^\alpha} + p = R'(\tilde{D})I^\beta. \tag{34}$$

If the value of ecosystem services for landowner is zero, the optimal outcome could be reached by setting $p = p_2 = \frac{E_s'(\tilde{N})}{I^\alpha} = R'(\tilde{D})I^\beta$. We can not say if p_2 is bigger or smaller than p_1 without specifying functions and parameter values. If $p_2 < p_1$, then setting the payment at the level p_1 will result in larger ecosystem area in equilibrium, $N > \tilde{N}$. Conversely, if $p_2 > p_1$, then $N < \tilde{N}$. Both results are not optimal in the model with infrastructure. But, as we can see, environmental consequences of decisions made on the basis of an incomplete model may vary depending on the parameters.
In the second case, the steady state is determined by the system of equations similar to (33):

$$\begin{cases} \frac{E'(N^*)}{(I^*)^\alpha} + p = R'(D^*)(I^*)^\beta \\ Q'(I^*) = \dfrac{\alpha E(N^*)}{(I^*)^{\alpha+1}} - \beta R(D^*)I^{\beta-1} \end{cases} \tag{35}$$

If the value of ecosystem services is not taken into account by the local goverment, the optimal foreign payment should be equal to the marginal private benefits $R'(D^*)(I^*)^\beta$ from development. In addition, if part of ecosystem services is external to the country (absorption of greenhouse gases by forests is the most obvious example), p should be set above the level of $R'(D)I^\beta$, to let the larger part of natural areas be preserved. Importantly, p will also affect the infrastructure density. The density of infrastructure in the steady state is expected to be lower, because the progressive expansion of the infrastructure will result in the reduction of ecosystem services. Thus, effective monitoring of payments for ecosystem services must also take into account the development of the infrastructure in the area concerned to avoid ecosystem services loss from fragmentation.

5 Conclusions

This work is an attempt to extend the basic competing land use model with an infrastructure factor which, according to many specialist studies, largely affects the quality of the environment and the range of available ecosystem services. The extended model leads to the conclusion that due to the dual impact of infrastructure (lowering the value of ecosystem services and increasing the private rents to developed land), the size of the natural area in the long-term equilibrium will be lower compared to the basic model. The preservation of nature ceases to be "profitable". Infrastructure also reduces the marginal costs of conversion, thus the converted area $c(t)$ is higher for every t. This result, although economically optimal, could be considered undesirable from the environmental point of view as threatening to sustainability.

If the decisions on optimal management of natural areas and infrastructure development are undertaken together, the result is a lower density of the infrastructure network and a larger ecosystem area. Lost ecosystem services due to increased fragmentation reduce the net benefits of infrastructure, thus the optimal density of infrastructure is lower.

Payments for ecosystem services were also considered within the discussed framework. As shown, the lack of consideration of the infrastructure effect in the basic model may lead to suboptimal development of the area, although this may have a different impact on the size of the preserved natural area depending on the parameters of the model.

The conclusions from the model should of course be subject to empirical research. It was found that in areas with more developed infrastructure, the conversion of natural areas occurs faster and on a larger scale. Although this statement can not be regarded as particularly revealing, it can be empirically tested using GIS. The results obtained can be used to more effectively protect natural areas. Secondly, it is worth considering the application of dynamic games theory to the issues of development and conversion of natural areas. It seems that this may be interesting in particular in relation to urban areas.

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Appendix 1

Let $E(N) = \omega\sqrt{N}$, $\quad R(D) = \sqrt{D}$, $\quad C(c) = \frac{1}{2}c^2$, where ω is a parameter representing the relative value of environmental benefits. Let $N_0 + D_0 = 100$, ie. total area. Then the necessary conditions (7) and (8) take form:

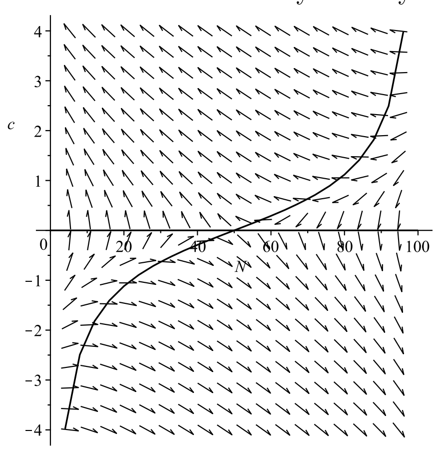
$$c = -\lambda, \qquad \rho\lambda - \dot{\lambda} = \frac{\omega}{\sqrt{N}} - \frac{1}{\sqrt{100 - N}}. \tag{36}$$

As $\dot{N} = -c$, we have the system of two ODE's:

$$\begin{cases} \dot{N} = -c \\ \dot{c} = \rho c + \dfrac{\omega}{\sqrt{N}} - \dfrac{1}{\sqrt{100 - N}} \end{cases} \tag{37}$$

with the stationary point $(N^*, c^*) = (100\dfrac{\omega^2}{1+\omega^2}, 0)$ – see Figure 1. Obviously, as ω goes to zero, ie. the value of ecosystem services is negligible to decision maker, the stationary point goes to $(0, 0)$ – the whole natural ecosystem is converted.

Figure 1: Phase portrait with nullclines for the dynamic system (37), $\omega = 1, \rho = 0.05$



Assuming $\omega = 1$, $\rho = 0.05$ the Jacobian for the system evaluated at stationary point $(50, 0)$ is

$$\begin{bmatrix} 0 & -1 \\ -\dfrac{\omega}{2N^{3/2}} - \dfrac{1}{2(100 - N)^{3/2}} & \rho \end{bmatrix} = \begin{bmatrix} 0 & -1 \\ -\dfrac{\sqrt{50}}{5000} & 0.05 \end{bmatrix}.$$

Both eigenvalues are real numbers with oposite sign, thus stationary point is a saddle. In general, for every $N_0 \in (0, 100)$, there is c_0 that the trajectory $(N(t), c(t))$ is a stable branch of the saddle path. If $N_0 > N^*$, then c_0 is positive and $c(t) \to 0$ as $t \to \infty$. For $N_0 < N^*$ the dynamics is analogous.