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Declaration of interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

CRediT authorship contribution statement

Charles Claron: Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing - Original draft, Writing - Review & editing, Visualisation; **Mehdi Mikou**: Conceptualisation, Methodology, Software, Formal analysis, Investigation, Data curation, Writing - Original draft, Visualisation; **Harold Levrel:** Conceptualisation, Methodology, Validation, Writing - Review & editing, Supervision, Project administration, Funding acquisition; **Léa Tardieu:** Conceptualisation, Methodology, Validation, Writing - Review & editing, Supervision, Project administration

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Highlights

- Conservation targeting improves the cost-effectiveness of incentive-based tools
- Return on investment analysis is used for a purchase of development rights program
- The program aims to limit the impact of soil sealing on ecosystem services supply
- Conservation benefits are measured via a synthetic urban ecosystem service index
- The approach is tested for spatial planning in the Greater Paris metropolis, France

Abstract

It is increasingly recognised that the good quality of life of city dwellers depends on the provision of a variety of urban ecosystem services (UES) within cities. However, soil sealing, associated with urbanisation and densification policies, affects soil multifunctionality and compromises the supply of future UES delivered both by public and private land. Incentive-based instruments could provide additional means of action for urban open space protection. Yet, their ability to produce conservation patterns that are cost-effective has been questioned, especially when they rely on public funding. To address this concern, this paper argues that conservation return on investment (ROI) analysis can be applied to UES supply protection objectives. We present an application of this method to a purchase of development right program within the Greater Paris metropolitan area (France). We assess and map the supply variation of three urban ecosystem services in case of soil sealing using Urban InVEST. These assessments are synthesised in the form of an index that serves as an indicator of the benefits of conservation investments. Conservation investment costs are based on estimates of the value of development rights from land market data. Finally, we use an expected-benefit-cost targeting strategy to produce maps showing the distribution of open land cells according to their regulatory status and their UES supply ROI. Our findings suggest that such maps provide a valuable decision-making tool to improve the cost-effectiveness of incentive-based conservation instruments and better inform land use decision planning.

Keywords

Purchase of development rights; Urban green space; Urban soils; Urban ecosystem services; Ecosystem services mapping; Cost-effectiveness

1. Introduction

Cities have an ambivalent relationship to ecosystem services (ES) (Daily 1997; MEA, 2005; Potschin-Young et al., 2018). The quality of life of city dwellers depends not only on the vast hinterlands that provide essential inputs of all kinds (Folke et al., 1997; Rees, 1992), but also on the proper functioning of urban ecosystems which supply a variety of *in situ* urban ecosystem services (UES) such as flood regulation, heat mitigation, or recreation (Bolund & Hunhammar, 1999; Gómez-Baggethun & Barton, 2013; Haase et al., 2014). Yet, urban development yields many threats to ecosystem health, mainly through land-use & land-cover changes and landscape fragmentation (Alberti, 2005; Cumming et al., 2014; Su et al., 2012). In particular, it is associated with soil sealing[3] which has been identified as the most acute form of soil degradation leading to a durable loss of ecosystem service intensity and diversity (O'Riordan et al., 2021; Pavao-Zuckerman & Pouyat, 2017; Tobias et al., 2018). In many established cities, this pressure on urban soil is very likely to increase due to densification policies implemented to limit urban sprawl and soil sealing outside city boundaries (Haaland and van den Bosch, 2015). Yet, this objective conflicts with the need to conserve urban green and blue infrastructure to meet the aspirations of urban dwellers and mitigate climate change (Elmqvist et al., 2015; Lwasa et al., 2022). In this context, tools to prioritise urban green spaces conservation are required.

The integration of UES into urban planning policies is widely seen as a critical element in improving the resilience of cities (Andersson et al., 2014; Artmann, 2014; Burkhard et al., 2014; Hansen & Pauleit, 2014; McPhearson et al., 2015; Niemelä et al., 2010; Tardieu et al., 2021; Teixeira da Silva et al., 2018). Despite uneven practical progress, this integration is now facilitated by sophisticated data and UES modelling devices, which allow to explore the potential of UES

[3] Soil sealing is defined as the destruction or covering of soils by buildings, constructions and layers of impermeable artificial material (Prokop et al., 2011). In simple terms it consists of covering the soil with an artificial impervious surface (Tobias et al., 2018).

mapping to inform urban and regional planning policies (Cimon-Morin & Poulin, 2018; Haase et al., 2014; Hamel et al., 2021; Lin et al., 2017; Rendon et al., 2019; Vollmer et al., 2016; Wei & Zhan, 2019). Moving from planning to action is a further challenge and city executives are in need of diversified and cost-effective policy instruments to implement spatial planning objectives (Gerber et al., 2018; Keeler et al., 2019). In this respect, incentive-based tools represent a promising opportunity: their role for habitat conservation is increasingly recognised by scientist and practitioners, especially on private land, yet they have rarely been applied to the preservation of UES (Cerra, 2017; Cortés Capano et al., 2019; Gooden & 't Sas-Rolfes, 2020; Polasky et al., 2011; Richards & Thompson, 2019).

Several governments have already invested significantly in such instruments, which has led to serious questions about their ability to direct investment to where they can produce an optimal conservation outcome to be raised (Arnold et al., 2013; Gooden & 't Sas-Rolfes, 2020; Kroeger & Casey, 2007; Parker & Thurman, 2019). Critics stress that incentive-based approaches to conservation fail in multiple instances due to imperfect and asymmetric information, imperfectly defined property rights and important transaction costs (Ferraro, 2008; Kroeger & Casey, 2007; Lockie, 2013; Vatn, 2015). Nevertheless, they have stimulated improvements in conservation planning methods such as conservation return-on-investment (ROI) analysis which aims at maximising a measure of conservation benefits generated by a limited budget (Boyd et al., 2015). To date, conservation ROI has mostly served biodiversity conservation purposes. If public investments in conservation are to be translated into UES preservation outcomes, ROI analysis needs to be adapted to this specific objective.

This article aims to address this implementation issue by asking to what extent ecosystem services modelling and mapping can inform the design of cost-effective incentive-based programs for UES supply preservation. We hypothesise that the recent development of ES evaluation methods and geographical information enables the production of decision support tools that help prioritise investment according to return for conservation of UES. To test this hypothesis, we present a methodological framework that applies a ROI analysis to the protection of 3 UES through a purchase of development rights program, in the Greater Paris metropolis (Section 3). Section 4 displays the priority investment maps that result from this protocol. The usefulness of this informational device is discussed in Section 5, as are the limitations of our methodology, before concluding remarks (Section 6). The following section provides some background information on conservation easements, their use in the purchase of development rights, and the relevance of conservation ROI analysis for their implementation.

2. Background

2.1 Incentive-based instruments: a growing solution for private land conservation

Nature conservation has traditionally relied on publicly managed networks of protected areas. Notwithstanding concerns about its efficiency, this strategy is now widely considered as insufficient to meet the global challenges of habitat conservation as it is designed to protect only specific and *remarkable* parts of natural environments (Butchart et al., 2015; Jones et al., 2018; Venter et al., 2018; Watson et al., 2014). Therefore, private land conservation (PLC) has emerged as a complementary strategy aimed at expanding space targeted for conservation by leveraging the contribution of private actors (Cortés Capano et al., 2019; Gooden & ’t Sas-Rolfes, 2020; Mitchell et al., 2018). It may resort to a wide array of instruments, but voluntary approaches, as opposed to regulatory or ‘command and control’ instruments, have been pivotal in the recent interest around PLC (Cortés Capano et al., 2019; Kamal et al., 2015). They rely on contractual or property rights tools, often combined with market mechanisms, monetary or non-monetary incentives (Doremus, 2003). Compared to command and control, they are presented as a flexible and efficient mean to correct habitat degradation (Armsworth et al., 2007; Engel et al., 2008; Perrings et al., 1992) and they circumvent the legal difficulties of property right infringement that often arise when land-use regulation diminishes land value (Lockie, 2013; Skuzinski & Linkous, 2018). This partly explains the craze for voluntary approaches in the USA from the 1980s onwards in a social context of distrust of state intervention (Kay, 2015; Rome, 2001).

Conservation easement is the PLC tool that has attracted the most attention from academics worldwide (Cortés Capano et al., 2019; Gooden & ’t Sas-Rolfes, 2020). It is a voluntary but legally binding agreement between an authorised organisation and a landowner to preserve the natural or heritage features of a piece of land (Cheever & McLaughlin, 2015; Kay, 2015; Parker, 2004). It is also a flexible tool: parties agree over the precise obligations (which can be positive or restrictive), contract duration (which can be perpetual in some jurisdictions) and the forms and amount of compensations devolved to the landowner (Boyd et al., 2000). It is finally a property right tool: the contractual commitment *runs with the land* and binds subsequent landowners throughout its duration (Parker & Thurman, 2019). Thus, compared to land acquisition, conservation easements provide comparable long-term protection at a lower cost, but at the expense of legal security as these contracts are subject to risks of release or breach (Fishburn et al., 2009; Hardy et al., 2017; Parker, 2004).

The basic principle behind this policy instrument originated in the United States in the 1930s with scenic easements designed to protect landscape integrity along highway infrastructure. From the 1960s onwards an increasing number of states adopted a specific legal status for "conservation easements". Promoted by federal enabling legislation, including tax deductions and uniformity doctrines, it quickly became a popular tool for habitat conservation (Buckland, 1987; Cheever &

McLaughlin, 2015; Kay, 2015). Across the USA, the share of conservation investment allocated through easements has been rising exponentially, exceeding 50% by 2003 (Fishburn et al., 2009), as a result they protect an area of 247,000 km² as of year-end 2020 (LTA, 2021). Based on this pioneering experience many countries have adopted legislation providing for conservation easement (Korngold, 2009; Račinska & Vahtrus, 2018), paving the way for the implementation of innovative incentive-based policy instruments.

2.2 The potential of purchase of development rights for urban ecosystem services preservation

Elaborating on the *property* as a *bundle of right* metaphor (Arnold, 2002; Galik & Jagger, 2015; Schlager & Ostrom, 1992), scholars have been presenting conservation easements as bargaining transactions allowing conservation organisations to acquire specific rights from a property title (Kay, 2015; Parker & Thurman, 2019). A purchase of development right (PDR) program is a voluntary policy instrument that compensates landowners willing to accept a conservation easement that restricts, often permanently, the development of their land in order to preserve its open space value. As they usually include restrictions over the development right, the phrases "conservation easement" and "purchase of development right" have sometimes been used interchangeably (Buckland, 1987; Daniels, 1991). However, conservation easements can, and increasingly do, impose more nuanced and diverse obligations affecting other rights of the *bundle*, for example when they require open access or mandate specific management actions (Rissman et al., 2013). Thus, PDR are a specific type of conservation easement whose sole object is to restrict a landowner's right to develop their land.

PDR programs have mainly been used in rural contexts for farmland and open space protection against urban sprawl (Bengston et al., 2004; Daniels, 1991). An early example is the Suffolk

County (State of New York, USA) PDR program for farmland preservation: adopted in 1974, it has protected almost 4,500 ha by the end of 2021 (Peterson & McCarthy, 1976; Lansdale, 2021). Although they rarely explicitly pursue this objective, the beneficial effects of PDR on ES conservation have been documented (Archibald et al., 2021; Benez-Secanho & Dwivedi, 2020; Crompton, 2009; Villamagna et al., 2015). As more and more cities consider using incentive tools to encourage the participation of private landowners in the conservation of UES, PDR could reveal itself to be a promising yet underused voluntary tool for the latter (Cerra, 2017; Morris, 2011; Richards & Thompson, 2019). Therefore, drawing on feedback from past experiences can be helpful in avoiding the pitfalls associated with its implementation.

2.3 Cost-effectiveness of conservation investments

Whether through direct monetary transfers or tax deductions, the growth of conservation easements and PDR programs worldwide has been supported by significant government expenditures, and as such have fallen under growing scrutiny (Vatn, 2015). Concerns have been raised about the ability of this incentive-based instrument to channel investments where they can create optimal conservation patterns (Gooden & 't Sas-Rolfes, 2020; Parker & Thurman, 2019). In the USA, critics focus on the dedicated generous tax relief system designed to foster easement adoption: it has proved complex to administer and has been widely misused for tax optimisation purposes (Looney, 2017; Rubin, 2017; Swift, 2010; Vercammen, 2018). This lack of demonstrated links between spendings and conservation outcomes is a common blind spot of incentive-based approaches in OECD countries (Hajkowicz, 2009).

The scientific literature has addressed this issue, showing that cost-effectiveness of these instruments can be improved by conservation return on investment (ROI) analysis (Boyd et al., 2015; Duke et al., 2013). Conservation ROI refers to *ex ante* analysis of investment opportunities to prioritise lands for conservation and maximise a measure of the benefits of – biodiversity or ES indicator – for a given budget (Boyd et al., 2015). This requires integrating economic costs into conservation planning in order to assess the performance of investments (Murdoch et al., 2007; Naidoo et al., 2006). Further proposals have been made to incorporate the likelihood (or risk) of land-use change to account for the actual threat to habitat conservation goals (Daniels, 1991; Newburn et al., 2005; Wilson et al., 2006). Approaches that combine measures of benefit, cost and risk associated with conservation investments have proven to yield higher performance in terms of ROI for conservation than any other targeting criteria (Boyd et al., 2015; Costello & Polasky, 2004; Newburn et al., 2006). ROI analysis has been applied to conservation easements and PDR programs, however most of these have relied on non-monetary biodiversity indicators to measure the benefits of conservation (Boyd et al., 2015). In the following sections we intend to tailor this framework to the specificities of UES conservation programs.

3. Materials and method

3.1 Overview and outline of the method section

This section presents the steps followed in this method section to apply ROI analysis to UES preservation. The goal is to provide conservation planners with a decision tool to prioritise the purchase of development rights among a set of parcels, given a limited budget. To do so, we rely on a static (one-time period) expected benefit-cost (EBC) targeting criterion which entails the computation and mapping of the following ratio for each land cells (equation 1) (Newburn et al., 2005, 2006):

$$EBC = \frac{Benefit\ of\ habitat\ conservation}{Cost\ of\ habitat\ conservation} \times risk\ of\ land\ conversion \tag{1}$$

We first present the study area and briefly discuss the applicability of purchase of development right (PDR) programs in this jurisdiction (section 3.2). We then lay out a 4-steps process for assessing and mapping the EBC (section 3.3). The first step (section 3.3.1) is to select a bundle of UES at stake with planning stakeholders and to identify the land cells suitable to provide these services. Step 2 displays our interpretation of the *risk of conversion* parameter. Based on zoning restrictions, targeted parcels are distributed between high (developable land) and low conversion risk (non-developable land) (section 3.3.2). In the third step we assess the benefits of conservation of each cell based on the calculation of a composite index that ranks them according to the decrease in their capacity to provide UES in case of sealing (section 3.3.3). Finally, we estimate the cost of restricting the development rights on these pieces of land through conservation easements (section 3.3.4).

3.2. Case study presentation

3.2.1 Study area: the Greater Paris metropolis

The Greater Paris metropolis [*Métropole du Grand Paris*], also called Greater Paris, is an intermunicipal administrative structure gathering 131 municipalities including Paris, the capital of France. The Greater Paris was established after two recent juridic acts, with the aim of developing a metropolitan project reconciling the improvement of the living environment of its inhabitants, the reduction of territorial inequalities and the development of a sustainable urban, social, and economic model[4]. It is located in the Ile-de-France region, in northern France and is divided into 12 territorial public bodies (sub-groups of communes) displayed in Figure 1 and referred to as

[4] The Law n° 2014-58 of January 27, 2014, and the Law n° 2015-991 of August 7, 2015. The general objectives assigned to this administrative unit are specified by article L5219-1 of the General Code of Territorial Communities [our translations].

"sub-territories" in the following. More detailed information on these sub-territories is provided in *Supplementary Information* (SI) Section 1.

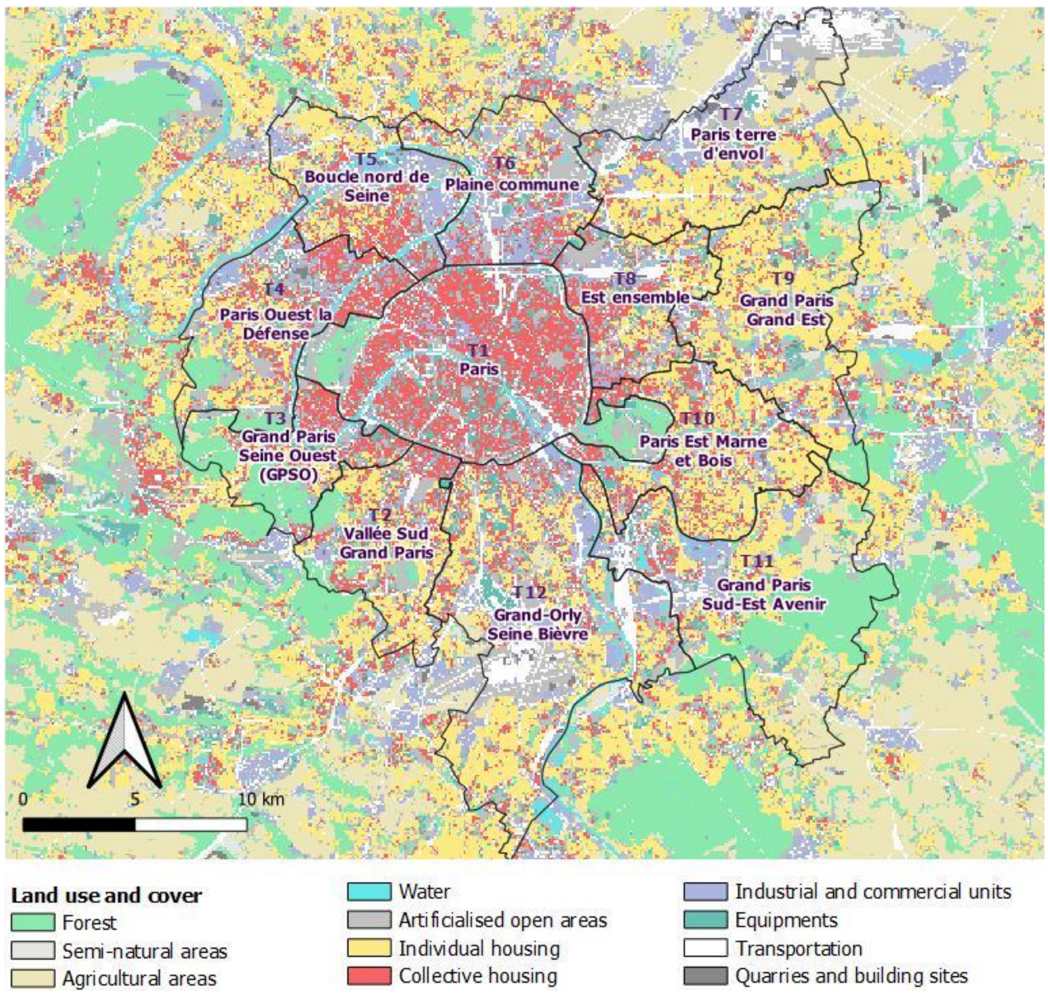

Figure 1: Land cover map of the greater Paris metropolis and its sub-territories

Note: The land-cover classes are based on LCM data which is further described in sub-section 3.3.1

In 2018, the Greater Paris housed around 7 million inhabitants within an area of 814 km² (INSEE, 2021), which makes it the most populated greater city in the European Union (Eurostat, 2021). Although it features a few parks, forests and gardens, it is a predominantly urbanised area with a population density of 8,689 per km² (INSEE, 2021). This density is expected to increase in the coming years as the Law n° 2010-597 of June 5, 2010, which plans the extension of the public transportation network, sets the objective of building 70,000 new housing units per year between 2010 and 2035. However, the modalities of this densification are being debated as several elected officials stress the need to preserve existing green infrastructures or to create new ones to help cope with the effects of climate change (Cazi, 2021).

3.2.2 Applicability of purchase of development right mechanisms to this area

The French equivalent to conservation easements (*obligation réelle environnementale)* has been adopted in 2016 and codified in article L132-3 of the Environment Code. The law specifies that the former must stipulate "reciprocal commitments", which may include financial compensation. Yet, unlike other countries such as the USA, French conservation easements cannot impose perpetual restrictions which are prohibited by the article 1210 of the Civil code. Therefore, the contract can only bind the parties for a maximum of 99 years. For the sake of simplicity in computation and generalisability of this empirical application, it is assumed that the conservator resorts to perpetual conservation easements.

Who could take on the role of conservator in this case? As conservation easements are fairly recent in France, there is not yet a structured governance around this instrument that would allow us to identify a relevant organisation to lead such a PDR program. However, in France, the implementation of public action is traditionally entrusted to state agencies or local authorities. We therefore assume that the conservator is a public body such as the Greater Paris metropolis itself, or the Ile-de-France region via its Green Spaces Agency.

In France, as in many Western countries, land development is regulated by zoning. Where development is allowed, a conservation easement can restrict the use of the development right. Where it is not, the situation is more nuanced: the right to build is no longer attached to the property title, but the owner remains free to encumber its land with an easement. In that case it functions as an additional protection, especially when future spatial planning changes are expected. The contract must therefore prohibit the current or future owner from applying for a building permit, should the land become developable. In this situation, the landowner is not giving up any right, therefore, formally speaking, such easement might be labelled as a *purchase of development option*. Having set out these legal subtleties, we will refer to both situations as PDR transactions.

3.3. Assessment of the expected benefit-cost ratio: a 4-step process

3.3.1. Step 1- Selection of the UES studied and identification of the service providing units

This research was conducted in the context of the IDEFESE participatory research project (https://idefese.wordpress.com/). We involved 56 stakeholders from the urban planning, environmental protection, NGOs and civil society sectors, representing more than 27 French and European institutions for the selection of UES and definition of indicators (Tardieu et al., 2021). These stakeholders selected three UES according to their perceived socioeconomic importance in the dense urban context of the Greater Paris: urban cooling (also called urban heat mitigation), flood risk mitigation and recreation.

In the context of ES assessment and mapping a service providing unit (SPU) is defined as “spatial unit within which an ecosystem service is provided” (Potschin-Young et al., 2018). In this study we divided the Greater Paris area into 100 x 100 m pixels, as most French planning and geographical statistics use the hectare as a unit of area. To identify the land cells suitable to provide these three UES – our SPUs – we used the most recent update of the *Land cover mode* (LCM) geographic database (Institut Paris Région, 2017). Based on aerial photography interpretation methods this data features a 25 x 25 m spatial resolution and consists of an inventory of land cover in 81 classes for its most detailed version – and 11 classes for its less subtle nomenclature. See SI, Section 2, for detailed nomenclature and correspondence between the 81 and 11-classes nomenclatures. We started by retaining non-sealed surfaces which correspond to items 1 to 28 of the 81-class nomenclature of LCM data. This choice is justified because soil sealing constitutes the most severe form of soil degradation, and because permeable surfaces are the parcels that might be adequately protected by a PDR program. We further excluded six land cover classes among this first selection. Four of them because they cannot be sealed and do not need to be protected: Closed waters [item 11], Watercourses [12], Swimming zones [20] and Cemeteries [26]. Two additional items have been discarded because we consider this type of land cover not suitable to provide the services under study: Intensive greenhouse cultivation [10] and Open tennis courts [19]. In the end, the 1ha pixels that feature a soil cover corresponding to one of the 22 remaining classes are the SPU that may be protected by a PDR program. The detailed list of these 22 land cover classes is presented in SI, Table S2.

3.3.2. Step 2- Risk of land-use change and soil sealing

France is a decentralised unitary state (Article 1 of the Constitution), where urban planning is mainly a municipal or intermunicipal jurisdiction. According to the article L151-9 of the urban planning code municipalities are responsible for drawing up the local land-use plans known as "*plan local d'urbanisme"* (PLU). This document classifies parcels into four zoning categories that regulate their development: urban (U), development (D), agricultural (A) and natural & forest (N) areas. Parcels in zones A and N cannot be developed, except in limited areas if building is justified by operational needs. Conversely, land in zones U or D are either urbanised or open to urban development. However, municipalities are at liberty to define sub-categories to tailor zoning regulation to contextual requirements. The geographical data about zoning regulations of the 131 communes of our study area is provided by the open-access dataset PLU Zonage (Apur, 2020). Land cells are distributed according to 11 zoning sub-categories: A, N, D and 8 urban classes (see SI, Section 3). However, for the purpose of this study, we only consider two urban subcategories: urban green areas (Ug) and other urban zones (Uo). Urban green area is a subcategory of zoning used by some municipalities to identify green spaces and urban recreation areas: parks, squares, sports facilities, cemeteries, etc. We consider that these zones are protected from development.

Our appreciation of land conversion risk is based on this zoning regulatory status. In the absence of empirical study about the probability of conversion of plots according to their zoning category, a simplification is made. We interpret the *likelihood of conversion* or *risk* component as a discrete variable taking two unobserved values: r_{nd}, risk of conversion of non-developable parcels (A, N, Ug); and r_d, the risk of conversion of developable parcels (Uo, D). We postulate the following relationship:

$$0 < r_{nd} < r_d < 1$$

Where r_{nd} is not a null probability since the regulatory status of such parcels can be changed to D through a modification of the PLU.

3.3.3. Step 3- Estimation of conservation benefits: a relative importance index

To compute the benefits of SPU conservation in terms of UES conservation we developed an index that synthesises information originating from three indicators of UES supply for each land pixel (Alam et al., 2016; Cortinovis et al., 2021; Hansen & Pauleit, 2014). This index is meant to assess the loss of the joint provision of the three UES under scrutiny in case of soil sealing of the SPUs. The capacity of a SPU to supply each of the three UES is assessed before and after a simulated soil sealing. Sealing was simulated for each SPU separately by changing its current land cover class to the class [34] (discontinuous collective housing), a building type that corresponds to contemporary constructions. Section 3.3.3.1 presents the three indicators of UES supply used in this study. Then, following the operation described in section 3.3.3.2 we synthesised the variation information of the three UES (difference before and after sealing) for each land-cell in a single index. This index enables to rank the different SPUs according to their conservation benefits: the higher the score of an SPU, the more its sealing would result in a loss of UES supply. To that extent it can be interpreted as a relative importance index (in UES provision), that helps setting conservation priorities. The spatial distribution of the variation of the three separate UES and the composite index can be found in the Supplementary Information, figure S4.

3.3.3.1 Assessment of the three UES with spatial modelling tools

Two prior considerations underlie our selection of UES indicators. First, we opted for non-monetary indicators, because we did not need to compare the benefits of conservation to an alternative option, as in a cost-benefit analysis. This was also a scientific orientation of the IDEFESE project, in line with the current position of the IPBES (Pascual et al., 2017). On the other hand, we focused on indicators of ES supply, defined as the potential of a particular spatial unit to provide a given service, regardless of its actual use (Potschin-Young et al., 2018). Table 1 displays the indicators that were retained by the scientific team in agreement with the stakeholders involved in the project. We describe them briefly, and further detail about their specificities is presented in SI, Section 4.

Table 1: Synthesis of urban ecosystem service supply indicators used in this study

UES	Indicator	Source
Urban cooling	Average cooling effect supplied by the pixel's vegetation within the Greater Paris	Urban InVEST 3.8 (Sharp et al., 2020)
Urban flood risk mitigation	Runoff retention volume in m^3 supplied by the pixel within each sub-watershed for a 30 mm rainfall depth	Urban InVEST 3.8 (Sharp et al., 2020)
Nature-based recreation	Number of potential beneficiaries of the pixel's recreational amenities, conditional on institutional and land cover parameters	In-house indicator

The assessment of urban cooling and urban flood mitigation services was realised using the output of the Integrated Valuation of ES and Tradeoffs (urban InVEST 3.8) software (Sharp et al., 2020) and has relied on the same parametrization used by Tardieu et al. (2021). The urban cooling service provided by an SPU is measured as the average cooling effect of vegetation throughout the Greater Paris. Flood risk mitigation is measured as the runoff retention capacity of the sub-watershed where a SPU is located for a given amount of rainfall.

The ability of a green space to provide recreational services depends not only on its natural characteristics but also on the social perceptions of its attributes, which may change over time and space (Scholte et al., 2018; Tardieu & Tuffery, 2019). For this UES we have created our own custom indicator consisting of two components. The firstis an institutional criterion of public accessibility. We relied on an expert-based analysis of LCM land-cover data to identify land-cover classes that are compatible with public access (Tardieu et al., 2021, SI, Section 3.8). It is assumed that only the land cells that meet this criterion can supply nature-based recreation. The second is a measure of potential accessibility: we estimated the number of beneficiaries within a disc-shape buffer around the SPU. To account for the fact that people are willing to travel various distances according to the type of green areas (Milcu et al., 2013), the radius of this buffer zone varies according to land cover classes. The parametrization of this variable was derived out of an empirical study of distance-based choice experiments in the study area (Ta et al., 2020). Equation 2 synthesises the computation of this indicator:

$$UES\_Recreation_{\,i} = \ 1\ Open\_Acces_i\ \times Pop\ in\ Radius_i \quad (2)$$

For a land cover of type i, the value of nature-based recreation depends on an indicator function that takes the value 1 if the land cover type entails open-access (0 otherwise) and considers the potential beneficiaries within variable radiuses. The values of these land-cover dependent parameters are detailed in SI, table S4.

3.3.3.2 Computing the relative importance index

Designing a composite index requires normalising, weighting, and aggregating pre-existent indicators (Gan et al., 2017; Pollesch & Dale, 2016). Normalisation consists in the adjustment of the observed variations for the ES provision indicators to a common and comparable scale (between 0 and 1), this was done using a Quantile Transformer with a uniform distribution (Pedregosa et al., 2011). We then opted for equal weighting, which means we adopted a neutral stance: each of the three UES is granted the same importance. Finally, aggregation was performed by summing the three measures of UES variations obtained in the previous step (equation 3). This additive aggregation method is the most straightforward and widely used in the scientific literature on sustainability indicators (Gan et al., 2017; Langhans et al., 2014). Unlike multiplicative methods it implies complete substitutability between different UES (Cortinovis et al., 2021). Multiplicative aggregation methods were not appropriate for our case-study. Indeed, as SPUs with private access do not provide any recreational UES, they show a null value for this ecosystem service which would imply a zero value of the composite index for many SPUs in a multiplicative framework.

$$Composite\ index = \Delta\,Recreation_{scaled} + \Delta\,Urban\_Flood_{scaled} + \Delta\,Urban\_Cooling_{scaled} \quad (3)$$

3.3.4 Step 4- Estimating conservation cost

This subsection presents the methodology used for calculating the cost of maintaining an UPR. We start by outlining the theoretical elements necessary to identify the different components of this cost (Section 3.3.4.1). We then present the data collected and used to estimate these costs in the case study (Section 3.3.4.2) and the assumptions made for calculating the cost of purchasing development rights by sub-territory (3.3.4.3).

3.3.4.1. Theoretical background: the cost of purchasing development rights

The cost of implementing a conservation instrument - such as PDR - consists of an opportunity cost and transaction costs (Boyd et al., 2000, 2015)**.** The latter depend primarily on the type of policy instrument at use. They can be divided between i) information: costs of acquiring necessary precontractual information; ii) contracting: costs related to the structuration of the transaction; and iii) enforcement cost: monitoring and ensuring that the terms of the agreement are satisfied throughout its duration (Boyd et al., 2000). In general, these costs are borne by the conservation organisation.

The opportunity cost is the loss associated with the alternative use of a property that society foregoes by protecting it (Potschin-Young et al., 2018). Put differently, it is the difference between private land value before and after the conservation restriction. This cost depends only on the nature of the restriction imposed; however, the choice of the instrument determines how it is shared among stakeholders (Lockie, 2013; Newburn et al., 2005). When development is prohibited by government regulation it is fully borne by the landowner, whereas if the government acquires the land it is borne by the taxpayer. PDR allows the sharing of this cost to be negotiated between the contractors. Depending on their respective willingness to pay and receive, the parties agree, among other things, on the amount of the monetary compensation paid by the conservator to the owner in exchange for the extinguishment of the development right. From the point of view of the conservator, this compensation can be labelled as the acquisition cost of the easement (Boyd et al., 2000, 2015).

As its goal is to devise the best possible use of a limited budget, the ROI framework ideally requires considering only the cost borne by the conservator: transaction and acquisition costs. However, the latter is difficult to estimate as there are no formal markets for easements or development rights where this price could be observed. Moreover, in the case of France, the number of conservation easements contracted is too small to provide any robust evidence (Claron, 2020). Therefore, a second-best option is to use opportunity cost as a proxy for acquisition cost (Boyd et al., 2015).

3.3.4.2. Purchase of development right cost component estimates

The estimates used to compute the cost of the PDR program, along with their respective reliability, are displayed at Table 2. Components of the transaction cost were estimated with semi-structured interviews of French conservation organisations and analysis of their activity reports (Claron, 2020). As we assumed the possibility of perpetual easement, the opportunity cost of a PDR conservation is equivalent to the market value of this right. When zoning restrictions are enforced, as in our study area, they can be used to reveal this value as the difference between the market-price of developable and non-developable vacant-land (Boyd et al., 2000; Daniels, 1991). To perform this estimation, we applied the protocol described in SI, Section 5, to the DVF+ dataset, which gathers real estate transactions over the last five years at the municipal level in France

(Cerema, 2021). Due to the insignificant number of appropriate transactions at the municipal level we performed this computation at sub-territory level.

Table 2: Estimates of the transaction and opportunity components of the conservation cost

Cost type		Estimates	Incurrence rule	Reliability
Transaction cost	Contracting	€1,000	At transaction time	Strong
	Information	€1,000	At transaction time	Low
	Enforcement	€500 per ha €20,000 per ha	Every year Net present cost	Medium
Opportunity cost		€6,196,786 per ha	At transaction time	Strong

For illustrative purposes, the value of the opportunity cost displayed in this table is the average value over the Greater Paris area. However, our estimation of PDR cost relied on estimates of this value at sub-territories scale which are presented in detail in SI, Table S8. Reliability comes from the authors' judgement. For enforcement cost, we indicate the net present cost of the annual expenditure of €500, based on a 2.5% discount rate.

3.3.4.3. Computation assumptions

Based on these estimates, the computation of the total cost of purchasing the development right of a 1ha SPU requires two additional assumptions. First, enforcement costs are incurred over multiple periods of time, for perpetuity, whereas information, contracting and acquisition costs are incurred immediately (Boyd et al. 2015). To convert these costs to a comparable timescale we calculated the net present value of enforcement costs, using the risk-free discount rate suggested for public investment planning by the French government's economic services: 2.5% (Quinet, 2013). Finally, we assumed that all SPUs bear the same transaction costs, and that acquisition costs amount to 100% of the development right value for developable land and 20% for non-developable land. In the first case, it seems reasonable to assume that landowners would at least expect to be compensated for the loss of development value. In contrast, for parcels already protected by zoning, the opportunity value claim may not be lawfully recognised in many jurisdictions (Boyd et al., 2015; Daniels, 1991). Yet, a minimal monetary compensation may be necessary to incentivise landowners to encumber their lands with an additional conservation easement. The 20% share was picked arbitrarily and does not impact our results since the benefit-cost ratios are computed in two distinct market segments: developable or not.

4. Results

4.1 Descriptive elements of the service providing units

Our selected SPUs represent 20,830ha (about 26% of the Great Paris area) and are unevenly distributed within the 12 sub-territories (Table 3). Five sub- territories (T1, T7, T9, T11 and T12) concentrate about 66% of the total SPUs. When we account for the size of each sub-territory relative to the Greater Paris area, some are overrepresented (T3, T7, T9 and T11) while the remaining ones are underrepresented.

Table 3 – Distribution of the service providing units according to the Greater Paris sub-territories

Sub-territory	Number of SPUs	Share of total SPUs	Area relative to Greater Paris area
T1	2,472	11.8%	12.9%
T2	1,148	5.5%	5.8%

T3	1,558	7.5%	4.5%
T4	1,445	6.9%	7.3%
T5	790	3.8%	6.1%
T6	876	4.2%	5.8%
T7	2,118	10.2%	9.6%
T8	518	2.5%	4.8%
T9	1,959	9.4%	8.8%
T10	644	3.1%	6.9%
T11	4,842	23.2%	12.3%
T12	2,460	11.8%	15.1%

In terms of zoning classes, more than 90% of the SPUs are in Natural (52%) or Urban areas (urban green: 6%; others: 34%). This unexpected high share of unsealed cells falling in the Urban zoning corresponds mainly to the following land cover classes: Parks or gardens [13], Outdoor sport fields [18] and Grassed surfaces with or without shrubs [27]. It seems that only a few municipalities are using Urban green subcategories for this purpose. Woods and forests [1] and Parks or gardens [13] are the main LCM categories falling under the Natural zoning (Figure 2).

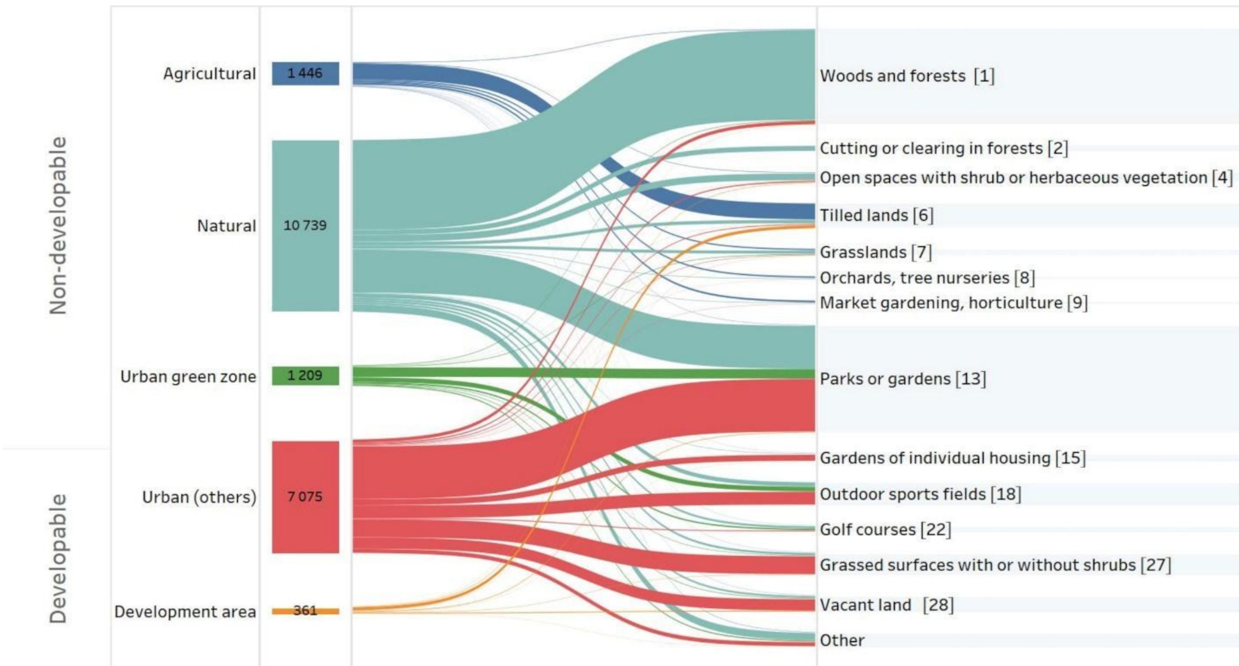

Figure 2 : Correspondence between PLU zoning and LCM categories

Note: As exposed in Section 3.3.2. developability is a variable based on the regulatory status (PLU zoning categories) of the SPUs. SPUs falling in underrepresented LCM categories are grouped under Other. Complete result table is displayed on SI Section 6.

4.2 Components of the expected benefit-cost approach

Figure 3 presents the distribution of the benefits of land conservation for each SPU according to its land cover and zoning class. SPUs falling in the forest LCM category appear to be non-developable while providing Medium, High, or Very High conservation benefits with an almost uniform distribution between these 3 classes. Cells within the Artificial open spaces category are mostly located (about 80%) in areas providing Very low or Low benefits of conservation. The Agricultural category seems to provide the least conservation for the three UES under scrutiny with 85% of cells situated in the Very low class of benefits. This is partly due to the high proportion of cells with private access leading to a null index of recreation within this category. See SI Section 6 for spatial and statistical distributions of the conservation benefits index.

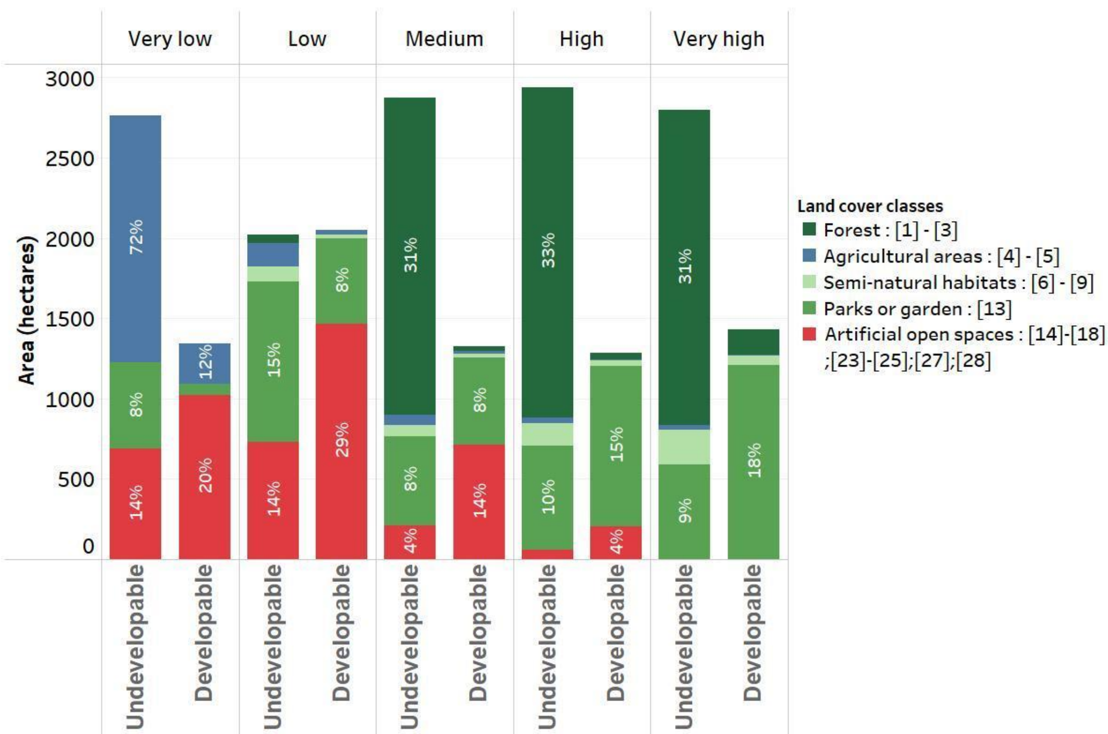

Figure 3: Benefits of land conservation for different LCM categories and PLU zoning types.

Note: For the sake of readability, the SPUs were divided into quintiles according to their benefits index. For this figure we relied on the 11-classes nomenclature of the LCM for land cover categories; the corresponding classes of the 81-item nomenclature are indicated via their numerical code in brackets (see SI Table S3). Since it is the most represented land cover category in our SPUs, we have nevertheless displayed 'parks and gardens' separately. Percentages represent the share of cells in each LCM category falling in one of the 5 categories of benefits of land conservation. For instance, forest lands are mainly (95%) non developable and feature 'medium', 'high' or 'very high' conservation benefits.

The costs of purchasing the development right over a 1ha land-cell are presented at Table 4. As the costs of protection of non-developable land are proportional to those of developable land, we limit our comments to the latter. Our results show significant geographical heterogeneity in protection costs. The net present costs show a standard deviation of €8,943,860 per ha, and a factor of 17 between the area with the highest protection cost (T1- City of Paris) and the one with the lowest (T9 - *Grand Paris Grand Est*). As the opportunity cost component is the only geography dependent variable, this discrepancy is only due to the observed difference in land market value.

Table 4 – Cost of conservation program for a 1ha land cell (SPU), by territorial public establishment and risk category

Sub - Territory	Developable		Non developable	
	Annual fee	Net present cost	Annual fee	Net present cost
T1	816,159	32,646,368	163,672	6,546,874
T2	211,262	8,450,462	42,692	1,707,692
T3	495,567	19,822,679	99,553	3,982,136
T4	473,086	18,923,446	95,057	3,802,289
T5	218,974	8,758,945	44,235	1,769,389
T6	145,223	5,808,928	29,485	1,179,386
T7	86,624	3,464,962	17,765	710,592
T8	138,690	5,547,595	28,178	1,127,119
T9	46,901	1,876,026	9,820	392,805
T10	95,544	3,821,745	19,549	781,949
T11	65,501	2,620,059	13,540	541,612
T12	147,284	5,891,356	29,897	1,195,871
GP	155,470	6,218,786	31,534	1,261,357

Note: *Costs are presented in two ways: as net present costs or as annualised costs for perpetuity computed with a discount rate of 2.5%.*

4.3 Benefit-cost ratio by risk categories

Figures 4 & 5 display the statistical and spatial distribution of the benefit-cost ratio within the Greater Paris metropolis. The composite index of conservation benefits has a lower dispersion than the conservation costs. The latter hence play a bigger role in the determination of the benefit-cost ratio than the benefits index. Like most urban areas, land values in the Greater Paris follow a centre-periphery gradient. It is also historically marked by a socioeconomic East-West divide: western areas concentrate higher land and property values (Clerval, 2022). Therefore, our results display a visible bias against protection in T1, T3 and T4 – where conservation costs are the highest – and pro conservation in T7, T9 and T11 – where they are the lowest. This also explains why territories with important conservation cost feature a lower variability in the benefit-cost index. Results for undevelopable areas show the presence of large land patches with similar values, they highlight the ability of the indicator to treat in almost the same way land parcels belonging to the same LCM category and in the same geographical unit. These coherent patches correspond to woods and forests: the municipal woods of the city of Paris (*Bois de Boulogne* & *Bois de Vincennes*) clearly appear.

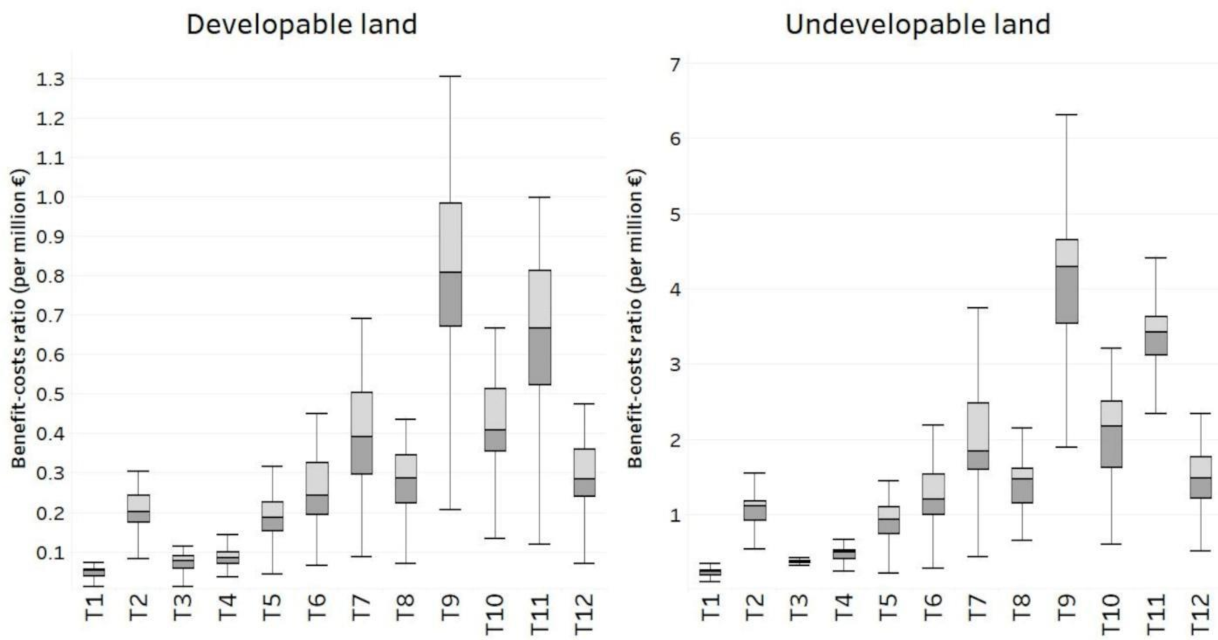

Figure 4: Statistical distribution of the benefit-cost index by risk classes at sub-territories level

Note: The difference in the scale of the y-axis between the two figures is due to the cost of conservation on non-developable plots being about 5-time lower (see Table 3).

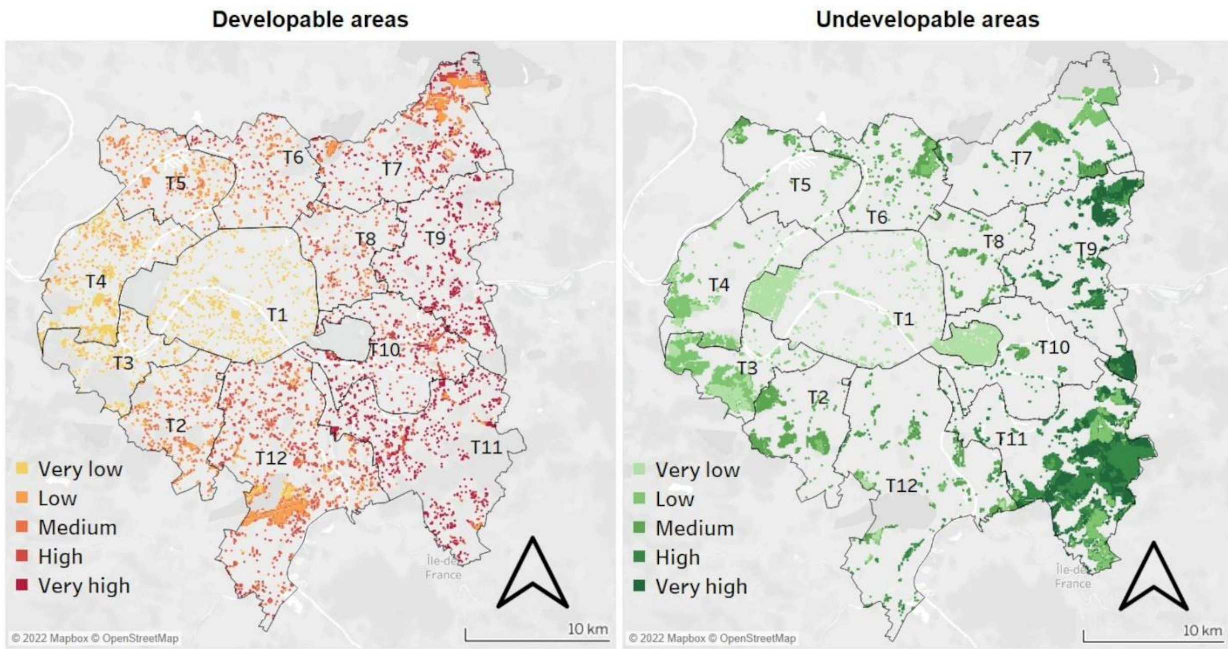

Figure 5: Priority investment areas by risk category

Note: For the sake of readability, the SPUs were divided into quintiles according to their benefit-cost ratio and ranked on a Likert scale

5. Discussion

5.1. Practical applications: a decision support tool

5.1.1. An input to the design of cost-effective PDR program

The maps presented in Figure 5 are strategic decision-support tools. They display the areas where public investment through development right purchase will generate the greatest conservation benefits in terms of UES supply. Such *ex-ante* analysis constitutes a valuable input to inform the practicalities of conservation transactions and improve the cost-effectiveness of PDR programs (Daniels, 1991).

First, it can help fine-tune contract design, which is an important element for reducing the cost of acquisition and fostering participation of private actors. Many PDR programs have been using reverse auction mechanisms to allocate funding, a system which encourages landowners to express their actual willingness to receive as well as their pro-social preferences (Ferraro, 2008; Liu, 2021; Lockie, 2013; Pascual & Perrings, 2007). Based on the maps, the program administrator can devise different auction mechanisms according to investment priority segment using, for instance, a multiple incentive system: offering tenders or fixed prices for high priority investment areas and using non-monetary incentives to enrol landowners of properties with low conservation ROI. Resorting to multiple incentive approaches reduces the possible crowding out effects of simple-incentive approaches and exploits the intrinsic motivations of participants for nature conservation (Beretti et al., 2019; Bowles, 2008; Cortés-Capano et al., 2021; Rode et al., 2015). This type of logic has been implemented by the government of New-South Wales (Australia): based on a 5-classes map of priority investment areas, landowners are offered diverse contracts with multiple incentives including voluntary uncompensated enrolment to conservation easements (OEH, 2018).

The decision support tool may also assist in addressing the spatial dimension of cost-effective conservation. Contiguity provides greater conservation benefits due to an *agglomeration effect*, which simple market mechanisms are often unable to take into account (Lynch & Liu, 2007). Among several corrective solutions, spatial targeting has proven its superiority to achieve better spatial coordination and conservation outcome (Fooks et al., 2016). Priority investment maps provide the program administrator with the appropriate information to favour contiguous plots in the purchasing process. In the hypothetical context of a reverse auction, the program administrator might increase the probability of acceptance of a bid conditional on the contiguity of a parcel to an already enrolled parcel or to other high-priority conservation areas (Daniels, 1991).

More generally, the ROI computation could be elaborated to include conservation bonuses for similar contiguity conditions.

5.1.2 An Information tool for an integrated UES conservation policy based on a diverse policy mix

So far, we have assumed a fixed conservation intensity implemented by a single instrument. While this assumption facilitates the estimation of costs, conservation in a complex socio-ecosystem such as the Greater Paris is a problem that requires a more diverse set of instruments. Since the map shows the areas where it is cost-efficient to buy development rights, it also underlines the areas where it is not and where other instruments may be more effective. Therefore, we suggest that this decision support tool can help planners articulate their instrument portfolio in support of UES conservation policy integration (Bengston et al., 2004; Capano & Howlett, 2020; Cejudo & Michel, 2021). The priority investment map could help set up market-based instruments such as *Transfer of development right* programs with value transfer from high to low conservation benefit areas. When protection is very costly, policy makers could opt for regulation instruments and change the status of land from developable to non-developable to reduce its conversion risk and safeguard its capacity to supply UES. In France, this means of action does not compel financial compensation but is prone to cause legal recourse.

Choosing the right instrument involves efficiency – instruments have varying transaction costs – and distributional issues – each instrument leads to a different sharing of opportunity costs (Curran et al., 2016; Fishburn et al., 2009). The question of whether the development right should be compensated, extinguished or transferred is a deliberative decision about who should bear the responsibility for protecting the land and what level of responsibility society may legitimately expect from landowners (Bromley & Hodge, 1990; Lockie, 2013; Moon et al., 2021).

5.2 Limitations

Some limitations in our data and methodology have consequences on the form and possible interpretations of these results. A first caveat lies in the simplistic interpretation of the conversion risk variable. The binary classification of plots according to their zoning class may be helpful from the point of view of decision-makers, but it could be refined to allow for greater spatial heterogeneity to be accounted for. In particular, two institutional factors may reduce the risk of land conversion, but were not included in the analysis due to the lack of available data. First, the ownership type of parcels, as public (state or communal) ownership is deemed to offer a higher degree of conservation of natural or forest areas (Kamal et al., 2015). Although we know that a significant portion of forests and natural areas are publicly owned in our study area, the lack of data on the type of ownership led us to assign the same risk factor to each undevelopable area. However, for example, the Bois de Boulogne or the Bois de Vincennes (in T1 - Paris) are municipal properties that are unlikely to be sealed. Secondly, besides urban zoning, other regulatory protections may apply to existing open land (e.g., protected areas) and reduce their risks of conversion. If the program were to be implemented by a public agency, it is likely that it would have the information enabling a more refined approach to the risk factor.

Our results are also limited by the spatial scale at which we calculated the conservation costs. Due to a limited number of land transactions we chose to assess the value of the development right at the sub-territory level. Consequently, the same value holds for the whole of Paris (T1), although it is characterised by significant spatial variability in property and land prices (Clerval, 2022). Improvements in this direction could be made by using more sophisticated estimation methods based on available property transaction data. In addition, it should be noted that the equivalence between opportunity cost and development right value is only valid if the easement is perpetual, as assumed in Section 3.2.2. For time-limited easements, like those in force in

French law, the opportunity cost might be lower than the value of the development rights since some future owner would eventually recover this right.

Results are sensitive to the synthesis approach of UES used in this case study. To build a synthetic index we have adopted an unweighted summation (equivalent to averaging) of our UES variations measures based on a preliminary standardisation. This method is common in the literature and adapted to the objectives of our study since we needed an indicator to measure the benefits of conservation (Cortinovis et al., 2021). However, it relies on a series of technical conventions that are not neutral. First the InVEST assessments models (and our custom recreation indicator) used in this study are based on many hypotheses that strongly influence the shape of UES variation measures (Sharp et al., 2020). Additionally, the method used to standardise the variation of individual UESdetermines the distribution, range, and dispersion of the synthetic indicator, which has at least two implications. First, alternative standardisation techniques might marginally change this distribution. Second, the relative scale and dispersion of the benefit indicator compared to those of costs (based on actual market data) are likely to influence the final distribution of the land cells within priority investment classes. As the range of benefit and cost measures are very different, we have conducted a robustness check and tried scaling our benefit measures to the same range as that of costs. As exposed in Section 7 of SI, more than 95% of the SPU stay in the same ROI category after this operation. Finally, the additive aggregation implies a hypothesis of perfect substitutability between UES that needs to be considered when interpreting the results. In this case, our composite index is based on the measurement of two regulating and one cultural ES, which are categories that interact synergistically (Maes et al., 2012; Nelson et al., 2009). If a wider range of UES were to be considered, this index might be less relevant as it would mask the trade-offs between services (Alam et al., 2016; Howe et al., 2014). In these conditions, other decision support tools should be implemented to help planners negotiate appropriate compromises (Boyd et al., 2015; Cortinovis et al., 2021).

Finally, the one-period static optimisation model presented in this study is based on a simplification of reality. Indeed, land value, development risk and relative importance in UES supply respond dynamically to conservation or development actions (Boyd et al., 2015; Costello & Polasky, 2004; Newburn et al., 2006). At the local level, the preservation of one parcel may drive-up both the price and conversion risk of neighbouring properties. At the metropolitan level, protecting a significant proportion of vacant land may increase land scarcity with effects on prices and conversion rates. In the meantime, the conversion of open land would affect the level of UES supplied by surroundings SPU. Therefore, conservators are rather faced with a problem of intertemporal optimisation of a limited budget over a given planning horizon. However, empirical comparisons reveal that while static approaches are somewhat less efficient, they offer significantly easier heuristics than dynamic stochastic intertemporal optimisation methods (Costello & Polasky, 2004; Meir et al., 2004; Wilson et al., 2006).

6. Conclusion

Cities are complex socio-ecosystems characterised by high population density and highly fragmented land tenure. Incentive-based instruments offer interesting and still underused opportunities to protect their ability to supply UES. However, past experiences with such instruments indicate that they may not be able to achieve their objectives fully without integrating appropriate targeting methods into their design principles. Using ES mapping models and socio-economic data, this paper demonstrates that ROI analysis can be applied to the design of a PDR program tailored for the safeguard of UES supply. Although targeting is a necessary step, it is only one of many dimensions that contribute to improving the cost-effectiveness of conservation policy instruments. Further developments could indicate how design principles at all stages of the policy instrument's life cycle (including evaluation) should be adapted to urban contexts. While additional improvements are required before this approach can be integrated in real-life instrument design, we proposed a proof of concept in a data-rich study area that could serve as an inspiration and be tested in other contexts.

This study focuses on improving the return on investment in UES supply preservation. Consequently, our results only constitute a partial decision support tool as they do not incorporate the analysis of UES demand, which constitutes a crucial aspect to link conservation planning to urban planning decisions that work for the common good. We suggest that future research should explore ways to incorporate UES supply and demand analysis (Cortinovis & Geneletti, 2020) to cost-effectiveness analysis. Finally, we would like to stress that the integration of ecosystem services into urban planning cannot be carried out on the sole basis of efficiency. It requires attention to environmental justice issues so that urban dwellers have access to sufficient ecosystem services to meet their basic needs (Liotta et al., 2020). Otherwise "(…) planning for multifunctionality might unintendedly increase environmental injustice for particular groups of society." (Hansen & Pauleit, 2014, p. 527).

Appendix A

Supplementary Information attached

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