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Mapping urban ecosystem services to design cost-effective purchase of development rights programs: the case of the Greater Paris metropolis

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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

2 Cities have an ambivalent relationship to ecosystem services (ES) (Daily 1997; MEA, 2005;
3 Potschin-Young et al., 2018). The quality of life of city dwellers depends not only on the vast
4 hinterlands that provide essential inputs of all kinds (Folke et al., 1997; Rees, 1992), but also on
5 the proper functioning of urban ecosystems which supply a variety of *in situ* urban ecosystem
6 services (UES) such as flood regulation, heat mitigation, or recreation (Bolund & Hunhammar,
7 1999; Gómez-Baggethun & Barton, 2013; Haase et al., 2014). Yet, urban development yields
8 many threats to ecosystem health, mainly through land-use & land-cover changes and landscape
9 fragmentation (Alberti, 2005; Cumming et al., 2014; Su et al., 2012). In particular, it is associated
10 with soil sealing³ which has been identified as the most acute form of soil degradation leading to
11 a durable loss of ecosystem service intensity and diversity (O’Riordan et al., 2021; Pavao-
12 Zuckerman & Pouyat, 2017; Tobias et al., 2018). In many established cities, this pressure on
13 urban soil is very likely to increase due to densification policies implemented to limit urban sprawl
14 and soil sealing outside city boundaries (Haaland and van den Bosch, 2015). Yet, this objective
15 conflicts with the need to conserve urban green and blue infrastructure to meet the aspirations of
16 urban dwellers and mitigate climate change (Elmqvist et al., 2015; Lwasa et al., 2022). In this
17 context, tools to prioritise urban green spaces conservation are required.

18 The integration of UES into urban planning policies is widely seen as a critical element in
19 improving the resilience of cities (Andersson et al., 2014; Artmann, 2014; Burkhard et al., 2014;
20 Hansen & Pauleit, 2014; McPhearson et al., 2015; Niemelä et al., 2010; Tardieu et al., 2021;
21 Teixeira da Silva et al., 2018). Despite uneven practical progress, this integration is now facilitated
22 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).

23 mapping to inform urban and regional planning policies (Cimon-Morin & Poulin, 2018; Haase et
24 al., 2014; Hamel et al., 2021; Lin et al., 2017; Rendon et al., 2019; Vollmer et al., 2016; Wei &
25 Zhan, 2019). Moving from planning to action is a further challenge and city executives are in need
26 of diversified and cost-effective policy instruments to implement spatial planning objectives
27 (Gerber et al., 2018; Keeler et al., 2019). In this respect, incentive-based tools represent a
28 promising opportunity: their role for habitat conservation is increasingly recognised by scientist
29 and practitioners, especially on private land, yet they have rarely been applied to the preservation
30 of UES (Cerra, 2017; Cortés Capano et al., 2019; Gooden & 't Sas-Rolfes, 2020; Polasky et al.,
31 2011; Richards & Thompson, 2019).

32
33 Several governments have already invested significantly in such instruments, which has led to
34 serious questions about their ability to direct investment to where they can produce an optimal
35 conservation outcome to be raised (Arnold et al., 2013; Gooden & 't Sas-Rolfes, 2020; Kroeger &
36 Casey, 2007; Parker & Thurman, 2019). Critics stress that incentive-based approaches to
37 conservation fail in multiple instances due to imperfect and asymmetric information, imperfectly
38 defined property rights and important transaction costs (Ferraro, 2008; Kroeger & Casey, 2007;
39 Lockie, 2013; Vatn, 2015). Nevertheless, they have stimulated improvements in conservation
40 planning methods such as conservation return-on-investment (ROI) analysis which aims at
41 maximising a measure of conservation benefits generated by a limited budget (Boyd et al., 2015).
42 To date, conservation ROI has mostly served biodiversity conservation purposes. If public
43 investments in conservation are to be translated into UES preservation outcomes, ROI analysis
44 needs to be adapted to this specific objective.

45
46 This article aims to address this implementation issue by asking to what extent ecosystem
47 services modelling and mapping can inform the design of cost-effective incentive-based programs

48 for UES supply preservation. We hypothesise that the recent development of ES evaluation
49 methods and geographical information enables the production of decision support tools that help
50 prioritise investment according to return for conservation of UES. To test this hypothesis, we
51 present a methodological framework that applies a ROI analysis to the protection of 3 UES
52 through a purchase of development rights program, in the Greater Paris metropolis (Section 3).
53 Section 4 displays the priority investment maps that result from this protocol. The usefulness of
54 this informational device is discussed in Section 5, as are the limitations of our methodology,
55 before concluding remarks (Section 6). The following section provides some background
56 information on conservation easements, their use in the purchase of development rights, and the
57 relevance of conservation ROI analysis for their implementation.

58 2. Background

59 2.1 Incentive-based instruments: a growing solution for private land 60 conservation

61 Nature conservation has traditionally relied on publicly managed networks of protected areas.
62 Notwithstanding concerns about its efficiency, this strategy is now widely considered as
63 insufficient to meet the global challenges of habitat conservation as it is designed to protect only
64 specific and *remarkable* parts of natural environments (Butchart et al., 2015; Jones et al., 2018;
65 Venter et al., 2018; Watson et al., 2014). Therefore, private land conservation (PLC) has emerged
66 as a complementary strategy aimed at expanding space targeted for conservation by leveraging
67 the contribution of private actors (Cortés Capano et al., 2019; Gooden & 't Sas-Rolfes, 2020;
68 Mitchell et al., 2018). It may resort to a wide array of instruments, but voluntary approaches, as
69 opposed to regulatory or 'command and control' instruments, have been pivotal in the recent

70 interest around PLC (Cortés Capano et al., 2019; Kamal et al., 2015). They rely on contractual or
71 property rights tools, often combined with market mechanisms, monetary or non-monetary
72 incentives (Doremus, 2003). Compared to command and control, they are presented as a flexible
73 and efficient mean to correct habitat degradation (Armsworth et al., 2007; Engel et al., 2008;
74 Perrings et al., 1992) and they circumvent the legal difficulties of property right infringement that
75 often arise when land-use regulation diminishes land value (Lockie, 2013; Skuzinski & Linkous,
76 2018). This partly explains the craze for voluntary approaches in the USA from the 1980s onwards
77 in a social context of distrust of state intervention (Kay, 2015; Rome, 2001).

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

90 The basic principle behind this policy instrument originated in the United States in the 1930s with
91 scenic easements designed to protect landscape integrity along highway infrastructure. From the
92 1960s onwards an increasing number of states adopted a specific legal status for "conservation
93 easements". Promoted by federal enabling legislation, including tax deductions and uniformity
94 doctrines, it quickly became a popular tool for habitat conservation (Buckland, 1987; Cheever &

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

101 2.2 The potential of purchase of development rights for urban 102 ecosystem services preservation

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

116
117 PDR programs have mainly been used in rural contexts for farmland and open space protection
118 against urban sprawl (Bengston et al., 2004; Daniels, 1991). An early example is the Suffolk

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

128 2.3 Cost-effectiveness of conservation investments

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

139 The scientific literature has addressed this issue, showing that cost-effectiveness of these
140 instruments can be improved by conservation return on investment (ROI) analysis (Boyd et al.,
141 2015; Duke et al., 2013). Conservation ROI refers to *ex ante* analysis of investment opportunities
142 to prioritise lands for conservation and maximise a measure of the benefits of – biodiversity or ES

143 indicator – for a given budget (Boyd et al., 2015). This requires integrating economic costs into
144 conservation planning in order to assess the performance of investments (Murdoch et al., 2007;
145 Naidoo et al., 2006). Further proposals have been made to incorporate the likelihood (or risk) of
146 land-use change to account for the actual threat to habitat conservation goals (Daniels, 1991;
147 Newburn et al., 2005; Wilson et al., 2006). Approaches that combine measures of benefit, cost
148 and risk associated with conservation investments have proven to yield higher performance in
149 terms of ROI for conservation than any other targeting criteria (Boyd et al., 2015; Costello &
150 Polasky, 2004; Newburn et al., 2006). ROI analysis has been applied to conservation easements
151 and PDR programs, however most of these have relied on non-monetary biodiversity indicators
152 to measure the benefits of conservation (Boyd et al., 2015). In the following sections we intend to
153 tailor this framework to the specificities of UES conservation programs.

154 3. Materials and method

155 3.1 Overview and outline of the method section

156 This section presents the steps followed in this method section to apply ROI analysis to UES
157 preservation. The goal is to provide conservation planners with a decision tool to prioritise the
158 purchase of development rights among a set of parcels, given a limited budget. To do so, we rely
159 on a static (one-time period) expected benefit-cost (EBC) targeting criterion which entails the
160 computation and mapping of the following ratio for each land cells (equation 1) (Newburn et al.,
161 2005, 2006):

162

$$EBC = \frac{\textit{Benefit of habitat conservation}}{\textit{Cost of habitat conservation}} \times \textit{risk of land conversion} \quad (1)$$

163

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

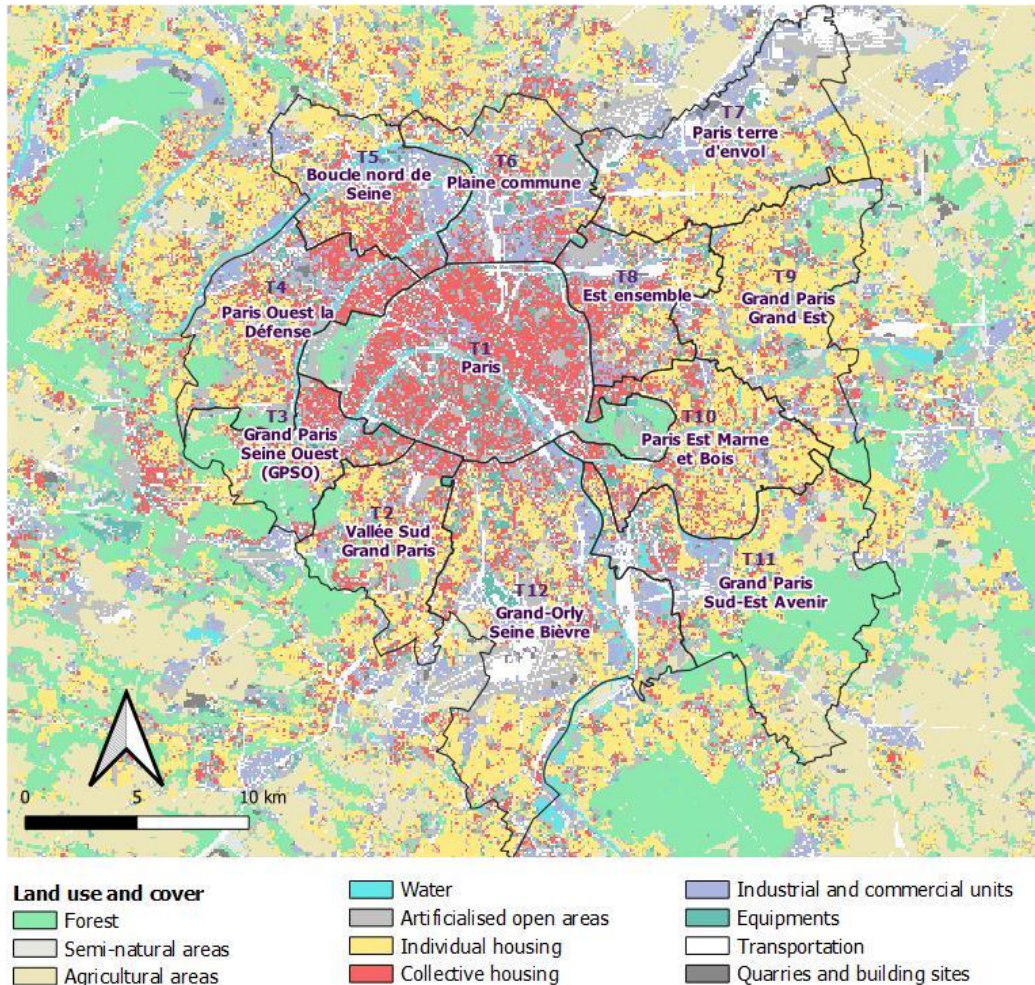
175 3.2. Case study presentation

176 3.2.1 Study area: the Greater Paris metropolis

177 The Greater Paris metropolis [*Métropole du Grand Paris*], also called Greater Paris, is an
178 intermunicipal administrative structure gathering 131 municipalities including Paris, the capital of
179 France. The Greater Paris was established after two recent juridic acts, with the aim of developing
180 a metropolitan project reconciling the improvement of the living environment of its inhabitants, the
181 reduction of territorial inequalities and the development of a sustainable urban, social, and
182 economic model⁴. It is located in the Ile-de-France region, in northern France and is divided into
183 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].

184 “sub-territories” in the following. More detailed information on these sub-territories is provided in
 185 *Supplementary Information (SI) Section 1.*
 186



187
 188 **Figure 1: Land cover map of the greater Paris metropolis and its sub-territories**
 189 *Note: The land-cover classes are based on LCM data which is further described in sub-section 3.3.1*

190 In 2018, the Greater Paris housed around 7 million inhabitants within an area of 814 km² (INSEE,
 191 2021), which makes it the most populated greater city in the European Union (Eurostat, 2021).
 192 Although it features a few parks, forests and gardens, it is a predominantly urbanised area with a
 193 population density of 8,689 per km² (INSEE, 2021). This density is expected to increase in the
 194 coming years as the Law n° 2010-597 of June 5, 2010, which plans the extension of the public

195 transportation network, sets the objective of building 70,000 new housing units per year between
196 2010 and 2035. However, the modalities of this densification are being debated as several elected
197 officials stress the need to preserve existing green infrastructures or to create new ones to help
198 cope with the effects of climate change (Cazi, 2021).

199 3.2.2 Applicability of purchase of development right mechanisms to this 200 area

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

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

215 In France, as in many Western countries, land development is regulated by zoning. Where
216 development is allowed, a conservation easement can restrict the use of the development right.
217 Where it is not, the situation is more nuanced: the right to build is no longer attached to the

218 property title, but the owner remains free to encumber its land with an easement. In that case it
219 functions as an additional protection, especially when future spatial planning changes are
220 expected. The contract must therefore prohibit the current or future owner from applying for a
221 building permit, should the land become developable. In this situation, the landowner is not giving
222 up any right, therefore, formally speaking, such easement might be labelled as a *purchase of*
223 *development option*. Having set out these legal subtleties, we will refer to both situations as PDR
224 transactions.

225 3.3. Assessment of the expected benefit-cost ratio: a 4-step 226 process

227 3.3.1. Step 1- Selection of the UES studied and identification of the service 228 providing units

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

236 In the context of ES assessment and mapping a service providing unit (SPU) is defined as “spatial
237 unit within which an ecosystem service is provided” (Potschin-Young et al., 2018). In this study
238 we divided the Greater Paris area into 100 x 100 m pixels, as most French planning and
239 geographical statistics use the hectare as a unit of area. To identify the land cells suitable to

240 provide these three UES – our SPUs – we used the most recent update of the *Land cover mode*
241 (LCM) geographic database (Institut Paris Région, 2017). Based on aerial photography
242 interpretation methods this data features a 25 x 25 m spatial resolution and consists of an
243 inventory of land cover in 81 classes for its most detailed version – and 11 classes for its less
244 subtle nomenclature. See SI, Section 2, for detailed nomenclature and correspondence between
245 the 81 and 11-classes nomenclatures. We started by retaining non-sealed surfaces which
246 correspond to items 1 to 28 of the 81-class nomenclature of LCM data. This choice is justified
247 because soil sealing constitutes the most severe form of soil degradation, and because
248 permeable surfaces are the parcels that might be adequately protected by a PDR program. We
249 further excluded six land cover classes among this first selection. Four of them because they
250 cannot be sealed and do not need to be protected: Closed waters [item 11], Watercourses [12],
251 Swimming zones [20] and Cemeteries [26]. Two additional items have been discarded because
252 we consider this type of land cover not suitable to provide the services under study: Intensive
253 greenhouse cultivation [10] and Open tennis courts [19]. In the end, the 1ha pixels that feature a
254 soil cover corresponding to one of the 22 remaining classes are the SPU that may be protected
255 by a PDR program. The detailed list of these 22 land cover classes is presented in SI, Table S2.

256 3.3.2. Step 2- Risk of land-use change and soil sealing

257 France is a decentralised unitary state (Article 1 of the Constitution), where urban planning is
258 mainly a municipal or intermunicipal jurisdiction. According to the article L151-9 of the urban
259 planning code municipalities are responsible for drawing up the local land-use plans known as
260 “*plan local d’urbanisme*” (PLU). This document classifies parcels into four zoning categories that
261 regulate their development: urban (U), development (D), agricultural (A) and natural & forest (N)
262 areas. Parcels in zones A and N cannot be developed, except in limited areas if building is justified
263 by operational needs. Conversely, land in zones U or D are either urbanised or open to urban
264 development. However, municipalities are at liberty to define sub-categories to tailor zoning

265 regulation to contextual requirements. The geographical data about zoning regulations of the 131
266 communes of our study area is provided by the open-access dataset PLU Zonage (Apur, 2020).
267 Land cells are distributed according to 11 zoning sub-categories: A, N, D and 8 urban classes
268 (see SI, Section 3). However, for the purpose of this study, we only consider two urban
269 subcategories: urban green areas (Ug) and other urban zones (Uo). Urban green area is a
270 subcategory of zoning used by some municipalities to identify green spaces and urban recreation
271 areas: parks, squares, sports facilities, cemeteries, etc. We consider that these zones are
272 protected from development.

273

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

280

281
282

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

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

285 3.3.3. Step 3- Estimation of conservation benefits: a relative importance

286 index

287 To compute the benefits of SPU conservation in terms of UES conservation we developed an
288 index that synthesises information originating from three indicators of UES supply for each land
289 pixel (Alam et al., 2016; Cortinovis et al., 2021; Hansen & Pauleit, 2014). This index is meant to

290 assess the loss of the joint provision of the three UES under scrutiny in case of soil sealing of the
291 SPUs. The capacity of a SPU to supply each of the three UES is assessed before and after a
292 simulated soil sealing. Sealing was simulated for each SPU separately by changing its current
293 land cover class to the class [34] (discontinuous collective housing), a building type that
294 corresponds to contemporary constructions. Section 3.3.3.1 presents the three indicators of UES
295 supply used in this study. Then, following the operation described in section 3.3.3.2 we
296 synthesised the variation information of the three UES (difference before and after sealing) for
297 each land-cell in a single index. This index enables to rank the different SPUs according to their
298 conservation benefits: the higher the score of an SPU, the more its sealing would result in a loss
299 of UES supply. To that extent it can be interpreted as a relative importance index (in UES
300 provision), that helps setting conservation priorities. The spatial distribution of the variation of the
301 three separate UES and the composite index can be found in the Supplementary Information,
302 figure S4.

303 3.3.3.1 Assessment of the three UES with spatial modelling tools

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

313

314 **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

315

316 The assessment of urban cooling and urban flood mitigation services was realised using the
 317 output of the Integrated Valuation of ES and Tradeoffs (urban InVEST 3.8) software (Sharp et al.,
 318 2020) and has relied on the same parametrization used by Tardieu et al. (2021). The urban
 319 cooling service provided by an SPU is measured as the average cooling effect of vegetation
 320 throughout the Greater Paris. Flood risk mitigation is measured as the runoff retention capacity of
 321 the sub-watershed where a SPU is located for a given amount of rainfall.

322

323 The ability of a green space to provide recreational services depends not only on its natural
 324 characteristics but also on the social perceptions of its attributes, which may change over time
 325 and space (Scholte et al., 2018; Tardieu & Tuffery, 2019). For this UES we have created our own
 326 custom indicator consisting of two components. The first is an institutional criterion of public
 327 accessibility. We relied on an expert-based analysis of LCM land-cover data to identify land-cover
 328 classes that are compatible with public access (Tardieu et al., 2021, SI, Section 3.8). It is assumed
 329 that only the land cells that meet this criterion can supply nature-based recreation. The second is
 330 a measure of potential accessibility: we estimated the number of beneficiaries within a disc-shape
 331 buffer around the SPU. To account for the fact that people are willing to travel various distances

332 according to the type of green areas (Milcu et al., 2013), the radius of this buffer zone varies
333 according to land cover classes. The parametrization of this variable was derived out of an
334 empirical study of distance-based choice experiments in the study area (Ta et al., 2020). Equation
335 2 synthesises the computation of this indicator:

336

$$337 \quad UES_{Recreation}_i = 1 \text{ Open_Acces}_i \times \text{Pop in Radius}_i \quad (2)$$

338

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

343 3.3.3.2 Computing the relative importance index

344 Designing a composite index requires normalising, weighting, and aggregating pre-existent
345 indicators (Gan et al., 2017; Pollesch & Dale, 2016). Normalisation consists in the adjustment of
346 the observed variations for the ES provision indicators to a common and comparable scale
347 (between 0 and 1), this was done using a Quantile Transformer with a uniform distribution
348 (Pedregosa et al., 2011). We then opted for equal weighting, which means we adopted a neutral
349 stance: each of the three UES is granted the same importance. Finally, aggregation was
350 performed by summing the three measures of UES variations obtained in the previous step
351 (equation 3). This additive aggregation method is the most straightforward and widely used in the
352 scientific literature on sustainability indicators (Gan et al., 2017; Langhans et al., 2014). Unlike
353 multiplicative methods it implies complete substitutability between different UES (Cortinovis et al.,
354 2021). Multiplicative aggregation methods were not appropriate for our case-study. Indeed, as
355 SPUs with private access do not provide any recreational UES, they show a null value for this

356 ecosystem service which would imply a zero value of the composite index for many SPUs in a
357 multiplicative framework.

358
359
$$\text{Composite index} = \Delta \text{Recreation}_{scaled} + \Delta \text{Urban_Flood}_{scaled} + \Delta \text{Urban_Cooling}_{scaled} \quad (3)$$

360 3.3.4 Step 4- Estimating conservation cost

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

366 3.3.4.1. Theoretical background: the cost of purchasing development rights

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

374 The opportunity cost is the loss associated with the alternative use of a property that society
375 foregoes by protecting it (Potschin-Young et al., 2018). Put differently, it is the difference between
376 private land value before and after the conservation restriction. This cost depends only on the
377 nature of the restriction imposed; however, the choice of the instrument determines how it is
378 shared among stakeholders (Lockie, 2013; Newburn et al., 2005). When development is

379 prohibited by government regulation it is fully borne by the landowner, whereas if the government
380 acquires the land it is borne by the taxpayer. PDR allows the sharing of this cost to be negotiated
381 between the contractors. Depending on their respective willingness to pay and receive, the parties
382 agree, among other things, on the amount of the monetary compensation paid by the conservator
383 to the owner in exchange for the extinguishment of the development right. From the point of view
384 of the conservator, this compensation can be labelled as the acquisition cost of the easement
385 (Boyd et al., 2000, 2015).

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

393 3.3.4.2. Purchase of development right cost component estimates

394 The estimates used to compute the cost of the PDR program, along with their respective reliability,
395 are displayed at Table 2. Components of the transaction cost were estimated with semi-structured
396 interviews of French conservation organisations and analysis of their activity reports (Claron,
397 2020). As we assumed the possibility of perpetual easement, the opportunity cost of a PDR
398 conservation is equivalent to the market value of this right. When zoning restrictions are enforced,
399 as in our study area, they can be used to reveal this value as the difference between the market-
400 price of developable and non-developable vacant-land (Boyd et al., 2000; Daniels, 1991). To
401 perform this estimation, we applied the protocol described in SI, Section 5, to the DVF+ dataset,
402 which gathers real estate transactions over the last five years at the municipal level in France

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

405

406 **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

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

412 3.3.4.3. Computation assumptions

413 Based on these estimates, the computation of the total cost of purchasing the development right
 414 of a 1ha SPU requires two additional assumptions. First, enforcement costs are incurred over
 415 multiple periods of time, for perpetuity, whereas information, contracting and acquisition costs are
 416 incurred immediately (Boyd et al. 2015). To convert these costs to a comparable timescale we
 417 calculated the net present value of enforcement costs, using the risk-free discount rate suggested
 418 for public investment planning by the French government's economic services: 2.5% (Quinet,
 419 2013). Finally, we assumed that all SPUs bear the same transaction costs, and that acquisition

420 costs amount to 100% of the development right value for developable land and 20% for non-
 421 developable land. In the first case, it seems reasonable to assume that landowners would at least
 422 expect to be compensated for the loss of development value. In contrast, for parcels already
 423 protected by zoning, the opportunity value claim may not be lawfully recognised in many
 424 jurisdictions (Boyd et al., 2015; Daniels, 1991). Yet, a minimal monetary compensation may be
 425 necessary to incentivise landowners to encumber their lands with an additional conservation
 426 easement. The 20% share was picked arbitrarily and does not impact our results since the benefit-
 427 cost ratios are computed in two distinct market segments: developable or not.

428 4. Results

429 4.1 Descriptive elements of the service providing units

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

435

436 **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%

437

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

444

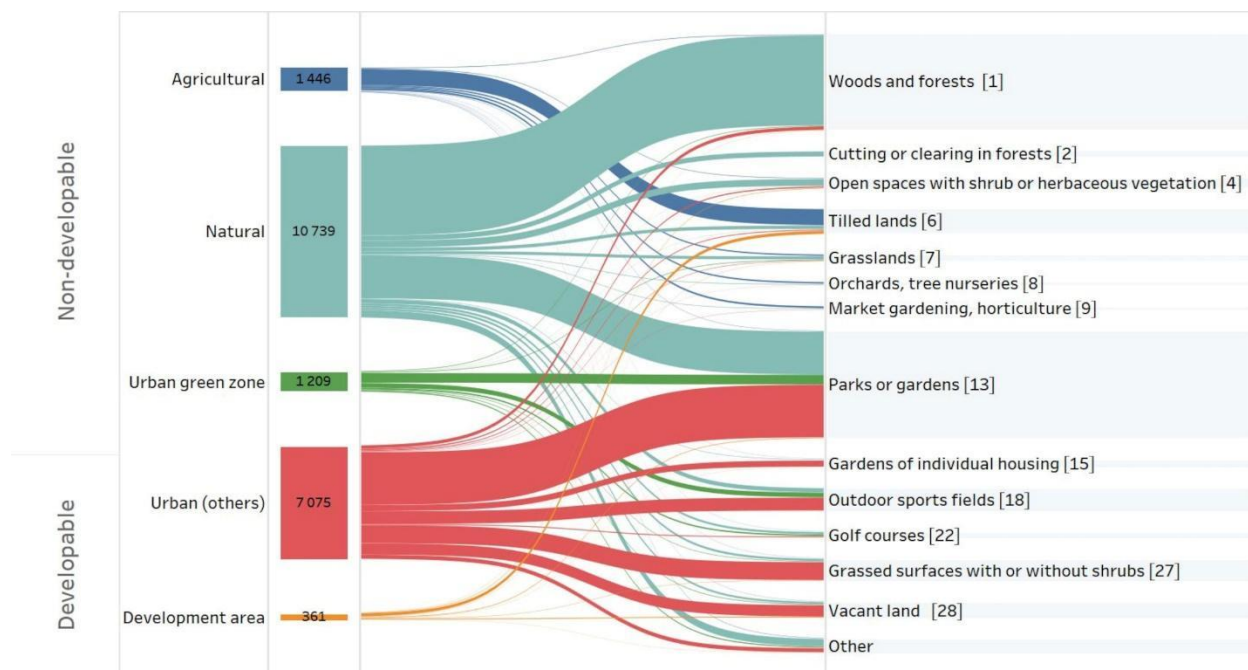


Figure 2 : Correspondence between PLU zoning and LCM categories

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.

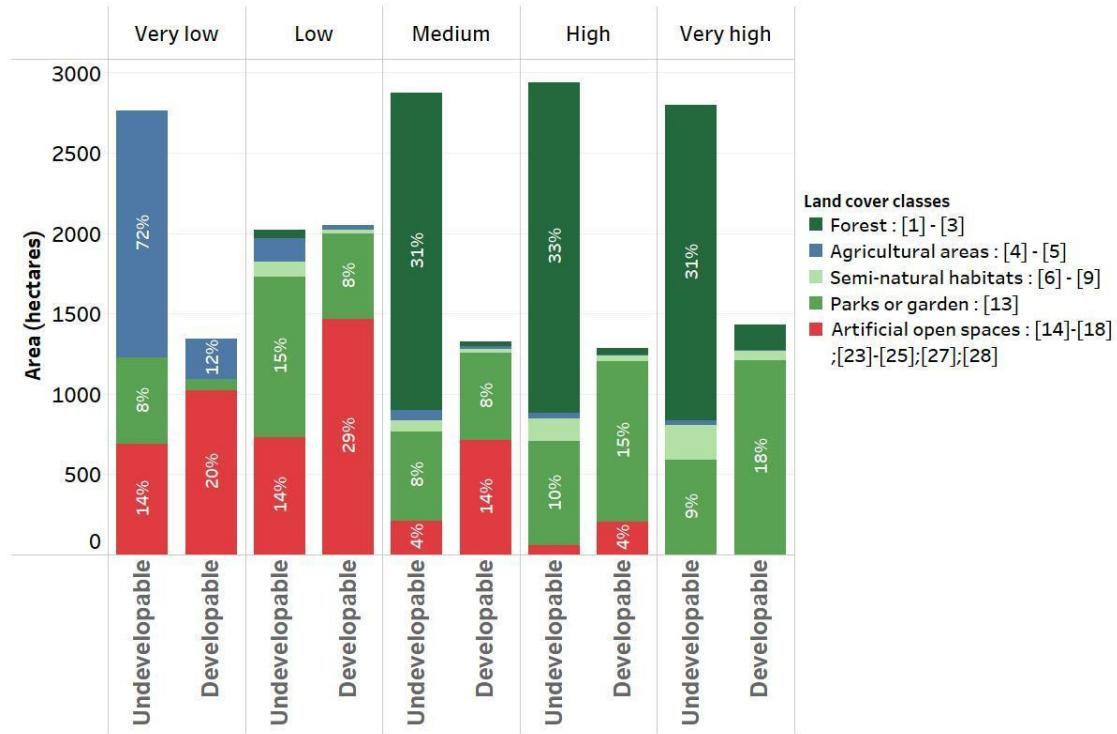


Figure 3: Benefits of land conservation for different LCM categories and PLU zoning types.

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

476 the lowest (T9 - *Grand Paris Grand Est*). As the opportunity cost component is the only geography
 477 dependent variable, this discrepancy is only due to the observed difference in land market value.
 478

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

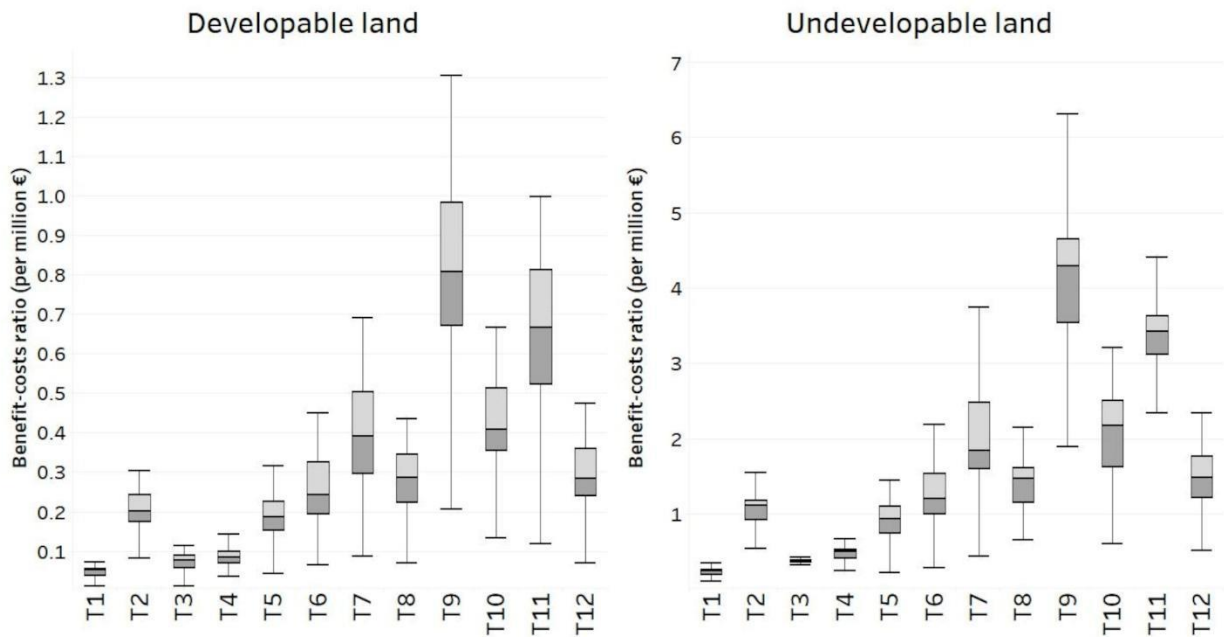
Sub - Territory	Developable		Non developable	
	Annual fee	Net present cost	Annual fee	Net present cost
T1	816,159	32,646,368	163,672	6,546,874
T2	211,262	8,450,462	42,692	1,707,692
T3	495,567	19,822,679	99,553	3,982,136
T4	473,086	18,923,446	95,057	3,802,289
T5	218,974	8,758,945	44,235	1,769,389
T6	145,223	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

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

483 4.3 Benefit-cost ratio by risk categories

484 Figures 4 & 5 display the statistical and spatial distribution of the benefit-cost ratio within the
 485 Greater Paris metropolis. The composite index of conservation benefits has a lower dispersion
 486 than the conservation costs. The latter hence play a bigger role in the determination of the benefit-

487 cost ratio than the benefits index. Like most urban areas, land values in the Greater Paris follow
 488 a centre-periphery gradient. It is also historically marked by a socioeconomic East-West divide:
 489 western areas concentrate higher land and property values (Clerval, 2022). Therefore, our results
 490 display a visible bias against protection in T1, T3 and T4 – where conservation costs are the
 491 highest – and pro conservation in T7, T9 and T11 – where they are the lowest. This also explains
 492 why territories with important conservation cost feature a lower variability in the benefit-cost index.
 493 Results for undevelopable areas show the presence of large land patches with similar values,
 494 they highlight the ability of the indicator to treat in almost the same way land parcels belonging to
 495 the same LCM category and in the same geographical unit. These coherent patches correspond
 496 to woods and forests: the municipal woods of the city of Paris (*Bois de Boulogne & Bois de*
 497 *Vincennes*) clearly appear.



498

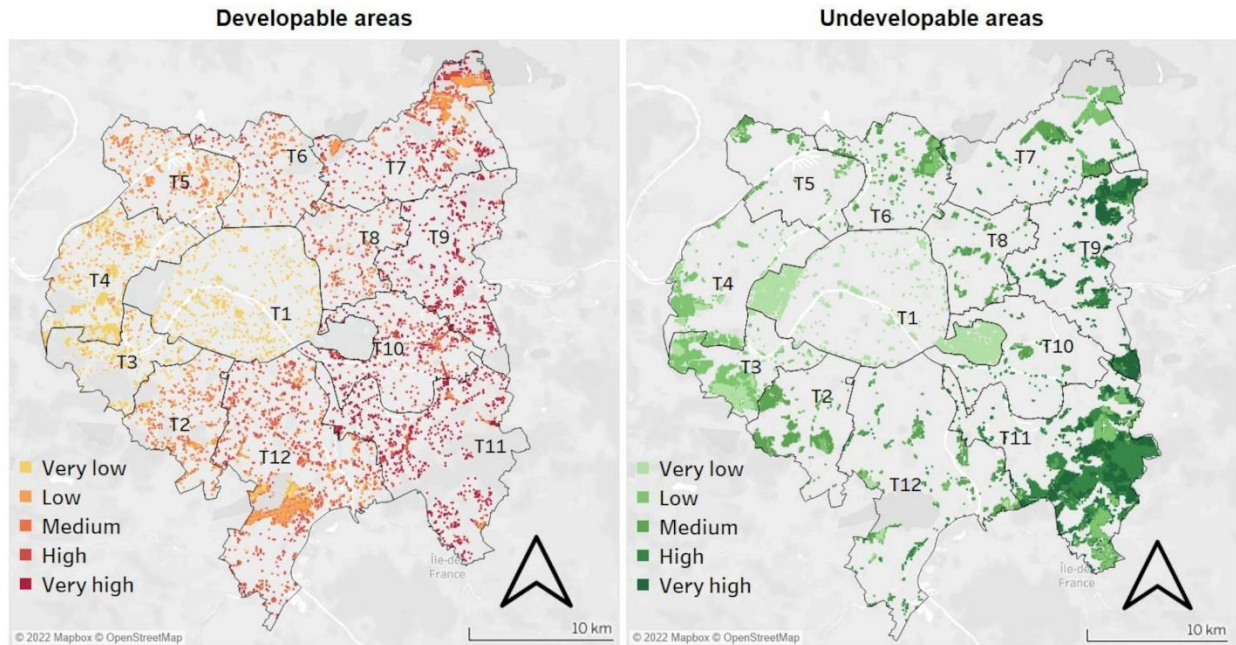
499

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

500

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).

501



502

503

Figure 5: Priority investment areas by risk category

504

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

505

on a Likert scale

506

5. Discussion

507

5.1. Practical applications: a decision support tool

508

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

509

The maps presented in Figure 5 are strategic decision-support tools. They display the areas

510

where public investment through development right purchase will generate the greatest

511

conservation benefits in terms of UES supply. Such *ex-ante* analysis constitutes a valuable input

512

to inform the practicalities of conservation transactions and improve the cost-effectiveness of PDR

513

programs (Daniels, 1991).

514 First, it can help fine-tune contract design, which is an important element for reducing the cost of
515 acquisition and fostering participation of private actors. Many PDR programs have been using
516 reverse auction mechanisms to allocate funding, a system which encourages landowners to
517 express their actual willingness to receive as well as their pro-social preferences (Ferraro, 2008;
518 Liu, 2021; Lockie, 2013; Pascual & Perrings, 2007). Based on the maps, the program
519 administrator can devise different auction mechanisms according to investment priority segment
520 using, for instance, a multiple incentive system: offering tenders or fixed prices for high priority
521 investment areas and using non-monetary incentives to enrol landowners of properties with low
522 conservation ROI. Resorting to multiple incentive approaches reduces the possible crowding out
523 effects of simple-incentive approaches and exploits the intrinsic motivations of participants for
524 nature conservation (Beretti et al., 2019; Bowles, 2008; Cortés-Capano et al., 2021; Rode et al.,
525 2015). This type of logic has been implemented by the government of New-South Wales
526 (Australia): based on a 5-classes map of priority investment areas, landowners are offered diverse
527 contracts with multiple incentives including voluntary uncompensated enrolment to conservation
528 easements (OEH, 2018).

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

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

540 5.1.2 An Information tool for an integrated UES conservation policy based 541 on a diverse policy mix

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

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

561 5.2 Limitations

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

577 Our results are also limited by the spatial scale at which we calculated the conservation costs.
578 Due to a limited number of land transactions we chose to assess the value of the development
579 right at the sub-territory level. Consequently, the same value holds for the whole of Paris (T1),
580 although it is characterised by significant spatial variability in property and land prices (Clerval,
581 2022). Improvements in this direction could be made by using more sophisticated estimation
582 methods based on available property transaction data. In addition, it should be noted that the
583 equivalence between opportunity cost and development right value is only valid if the easement
584 is perpetual, as assumed in Section 3.2.2. For time-limited easements, like those in force in

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

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

609 implemented to help planners negotiate appropriate compromises (Boyd et al., 2015; Cortinovis
610 et al., 2021).

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

623 6. Conclusion

624 Cities are complex socio-ecosystems characterised by high population density and highly
625 fragmented land tenure. Incentive-based instruments offer interesting and still underused
626 opportunities to protect their ability to supply UES. However, past experiences with such
627 instruments indicate that they may not be able to achieve their objectives fully without integrating
628 appropriate targeting methods into their design principles. Using ES mapping models and socio-
629 economic data, this paper demonstrates that ROI analysis can be applied to the design of a PDR
630 program tailored for the safeguard of UES supply. Although targeting is a necessary step, it is
631 only one of many dimensions that contribute to improving the cost-effectiveness of conservation

632 policy instruments. Further developments could indicate how design principles at all stages of the
633 policy instrument's life cycle (including evaluation) should be adapted to urban contexts. While
634 additional improvements are required before this approach can be integrated in real-life
635 instrument design, we proposed a proof of concept in a data-rich study area that could serve as
636 an inspiration and be tested in other contexts.

637

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

649 Appendix A

650 Supplementary Information attached

651

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