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Landscape Ecology

The influence of landscape spatial configuration on nitrogen and phosphorus exports in agricultural catchments --Manuscript Draft--

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Corresponding Author:	Antoine Casquin, M.S. Institut National de Recherche pour l'Agriculture l'Alimentation et l'Environnement Rennes, Bretagne FRANCE	
Corresponding Author Secondary Information:		
Corresponding Author's Institution:	Institut National de Recherche pour l'Agriculture l'Alimentation et l'Environnement	
Corresponding Author's Secondary Institution:		
First Author:	Antoine Casquin, M.S.	
First Author Secondary Information:		
Order of Authors:	Antoine Casquin, M.S. Rémi Dupas, PhD Sen Gu Ewan Couic, PhD Gérard Gruau, PhD Patrick Durand, PhD	
Order of Authors Secondary Information:		
Funding Information:	Interreg (Channel Payments for Ecosystem Services)	Dr. Rémi Dupas Dr Gérard Gruau Dr Patrick Durand
Abstract:	<p>Nitrogen (N) and phosphorus (P) exports from rural landscapes can cause eutrophication of inland and coastal waters. Few studies have investigated the influence of the spatial configuration of nutrient sources – i.e. the spatial arrangement of agricultural fields in headwater catchments – on N and P exports.</p> <p>This study aimed to 1) assess the influence of the spatial configuration of nutrient sources on nitrate (NO₃⁻) and total phosphorus (TP) exports at the catchment scale, and 2) investigate how relationships between landscape composition and landscape configuration vary depending on catchment size.</p> <p>We analysed NO₃⁻ and TP in 19 headwaters (1-14 km², Western France) every two weeks for 17 months. The headwater catchments had similar soil types, climate, and farming systems but differed in landscape composition (percentage agricultural land-use) and spatial configuration. We developed a landscape configuration index (LCI) describing the spatial configuration of nutrient sources as a function of their hydrological distance to streams and flow accumulation zones. We calibrated the LCI's two parameters to maximise the rank correlation with median concentrations of TP and NO₃⁻.</p> <p>We found that NO₃⁻ exports were controlled by landscape composition, whereas TP exports were controlled by landscape configuration. For a given landscape composition, landscape spatial configuration was heterogeneous at small scales (<10 km²) but homogeneous at larger scales (>50 km²).</p>	

	<p>The spatial configuration of nutrient sources influences TP but not NO₃- exports. An ideal placement of mitigation options to limit TP export should consider both the hydrological distance to streams and flow accumulation zones.</p>
<p>Suggested Reviewers:</p>	<p>Sarah E Godsey, PhD PI, Idaho State University godsey@isu.edu Relevant work on landscape pattern and biogeochemistry of headwater streams</p> <hr/> <p>Erin Peterson, PhD PI, Queensland University of Technology erin.peterson@qut.edu.au Research Interests: Environmental monitoring assessment and reporting Spatial networks Spatial statistics</p> <p>Her work on spatially explicit metrics and water quality</p> <hr/> <p>Jay Zarnetske, PhD Associate professor, Michigan State University jtz@msu.edu Research interest : how catchment hydrology and groundwater - surface water interactions affect aquatic ecosystems through solute generation, fate, and transport</p> <hr/> <p>Ian A Thomas, PhD Post Doc, University College Dublin ian.thomas@ucd.ie Specialist of critical source areas ;agricultural diffuse pollution; hydrological modelling</p> <hr/> <p>Philip Haygarth, PhD Professor, Lancaster Environment Centre p.haygarth@lancaster.ac.uk Professor Phil Haygarth conducts research on the interface between soils and freshwaters, with a focus on diffuse (particularly phosphorus) pollution and runoff control in a catchment context.</p> <p>Numerous works on scaling the phosphorus signal from land to aquatic ecosystems.</p>

[Click here to view linked References](#)

1 **Title**

2 The influence of landscape spatial configuration on nitrogen and phosphorus exports in
3 agricultural catchments

4 **Authors**

5 Antoine Casquin. INRAE, UMR SAS 1069 / L'Institut Agro, 65 rue de St Brieuc, 35200
6 Rennes, France. antoine.casquin@inrae.fr / a.casquin@gmail.com. [https://orcid.org/0000-](https://orcid.org/0000-0001-6897-0183)
7 0001-6897-0183

8 Rémi Dupas. INRAE, UMR SAS 1069 / L'Institut Agro, 65 rue de St Brieuc, 35200 Rennes,
9 France. remi.dupas@inrae.fr. <https://orcid.org/0000-0002-5932-4056>

10 Sen Gu. OSUR, Géosciences Rennes, CNRS, UMR 6118, Campus de Beaulieu, 35042
11 Rennes, France / Institute of Hydrobiology, Chinese Academy of Sciences, Wuhan 430072,
12 China. gusen@ihb.ac.cn

13 Ewan Couic. OSUR, Géosciences Rennes, CNRS, UMR 6118, Campus de Beaulieu, 35042
14 Rennes, France. ewan.couic@univ-rennes1.fr. <https://orcid.org/0000-0003-4223-2726>

15 Gérard Gruau. OSUR, Géosciences Rennes, CNRS, UMR 6118, Campus de Beaulieu,
16 35042 Rennes, France. gerard.gruau@univ-rennes1.fr

17 Patrick Durand. INRAE, UMR SAS 1069 / L'Institut Agro, 65 rue de St Brieuc, 35200
18 Rennes, France. patrick.durand@inrae.fr. <https://orcid.org/0000-0002-0984-693X>

19 **Correspondent author:**

20 Antoine Casquin. INRAE, UMR SAS 1069 / L'Institut Agro, 65 rue de St Brieuc, 35200
21 Rennes, France. antoine.casquin@inrae.fr (institutional) / a.casquin@gmail.com
22 (permanent). Tel.: +33(0)2 23 48 54 22. Fax: + 33(0)2 23 48 54 30.

23 **Abstract**

24 **Context**

25 Nitrogen (N) and phosphorus (P) exports from rural landscapes can cause eutrophication of
26 inland and coastal waters. Few studies have investigated the influence of the spatial
27 configuration of nutrient sources – i.e. the spatial arrangement of agricultural fields in
28 headwater catchments – on N and P exports.

29 **Objectives**

30 This study aimed to 1) assess the influence of the spatial configuration of nutrient sources on
31 nitrate (NO₃⁻) and total phosphorus (TP) exports at the catchment scale, and 2) investigate

32 how relationships between landscape composition and landscape configuration vary
33 depending on catchment size.

34 **Methods**

35 We analysed NO_3^- and TP in 19 headwaters (1-14 km^2 , Western France) every two weeks for
36 17 months. The headwater catchments had similar soil types, climate, and farming systems
37 but differed in landscape composition (percentage agricultural land-use) and spatial
38 configuration. We developed a landscape configuration index (LCI) describing the spatial
39 configuration of nutrient sources as a function of their hydrological distance to streams and
40 flow accumulation zones. We calibrated the LCI's two parameters to maximise the rank
41 correlation with median concentrations of TP and NO_3^- .

42 **Results**

43 We found that NO_3^- exports were controlled by landscape composition, whereas TP exports
44 were controlled by landscape configuration. For a given landscape composition, landscape
45 spatial configuration was highly heterogeneous at small scales (<10 km^2) but became
46 homogeneous at larger scales (>50 km^2).

47 **Conclusions**

48 The spatial configuration of nutrient sources influences TP but not NO_3^- exports. An ideal
49 placement of mitigation options to limit TP export should consider both the hydrological
50 distance to streams and flow accumulation zones

51

52 **Keywords**

53 Eutrophication; landscape pattern; landscape organisation; landscape composition;
54 heterogeneity; scaling

55

56 **1. Introduction**

57 Excessive loads of nitrogen (N) and phosphorus (P) cause the eutrophication of marine and
58 freshwater ecosystems (Dodds and Smith 2016; Le Moal et al. 2018), which threatens
59 biodiversity and human activities (Steffen et al. 2015). In streams and rivers that drain
60 agricultural landscapes, N and P originate mainly from fertilised agricultural fields. Water
61 quality mitigation measures often result in reduced yields or increased production costs;
62 hence, they decrease profitability for farmers (Withers et al. 2014). It is therefore necessary
63 to locate mitigation options such as set-aside areas and buffer zones where they will be most
64 effective (Cole et al. 2020). Mechanistic models (Casal et al. 2019; McDowell et al. 2014)
65 and methods based on statistical analysis of landscape properties (Doody et al. 2012;
66 Hashemi et al. 2018) have been used to optimise locations of mitigation practices.

67 Most N and P enter the hydrographic network in headwater streams (Dodds and Oakes
68 2008), which represent 90% of global stream length (Downing 2012) but are rarely
69 monitored, except for research purposes (Bishop et al. 2008). At this scale, nutrient exports
70 vary greatly in space (Abbott et al. 2018; Temnerud and Bishop 2005). While in large (>100
71 km²) catchments, both N and P exports can be statistically related to proxies of agricultural
72 pressure intensity, such as percentage of agricultural land-use or agricultural surplus (Dupas
73 et al. 2015; Goyette et al. 2018), these relationships break down at the scale of headwater
74 catchments (<20 km²) (Bol et al. 2018; Burt and Pinay 2005). The reason why the
75 relationship between landscape composition and nutrient exports is scale-dependent is
76 unclear. Identifying the factor(s) responsible for this loss of relationship, and the
77 characteristic spatial scale at which it occurs, would help guide water quality mitigation
78 measures at the catchment scale. In this article, we explore the hypothesis that the spatial
79 configuration of nutrient sources – i.e. the spatial arrangement/distribution/organisation of
80 agricultural fields, hereafter called “landscape configuration”, –influences N and P exports at
81 the headwater catchment scale. For a given region, landscape configuration can be highly
82 heterogeneous among headwater catchments but homogeneous among larger catchments
83 (Abbott et al. 2018; Temnerud and Bishop 2005). If verified, these two hypotheses could
84 explain the breakdown of the relationship between agricultural pressure intensity and nutrient
85 exports at the headwater catchment scale.

86 Several authors have reviewed the influence of landscape spatial configuration on nutrient
87 loads at multiple scales (Lintern et al. 2018; Uemaa et al. 2007). Metrics used in landscape
88 ecology are based on the area/density/edge, shape, isolation, interspersion, and connectivity
89 of patches, and they have been applied to predict water quality parameters at the catchment
90 scale (Liu et al. 2020; Shi et al. 2013; Xiao et al. 2016). However, these approaches often
91 rely on regressions of several landscape metrics and water quality parameters, which risks

92 over-fitting certain relationships and/or spurious correlations. Landscape spatial configuration
93 can also be understood as the overlap and proximity of features such as streams,
94 depressions, and flow accumulation areas to land-use patches (e.g. crops, forest fragments,
95 urban fabric). In line with this concept, Peterson et al. (2011)) related parameters of stream
96 ecological conditions to multiple spatially explicit landscape “topological” metrics and found
97 that these spatially explicit methods clearly had more predictive power than landscape
98 composition metrics. Staponites et al. (2019)), using similar metrics, suggested that the
99 spatial organisation of nutrient sources (i.e. landscape configuration) influenced the transfer
100 of reactive or labile water quality parameters (e.g. TP, NO_2^- and PO_4^{3-}), whereas percentages
101 of land use (i.e. landscape composition) have more predictive power for more temporally
102 stable water quality parameters (e.g. Ca^{2+} , NO_3^- , conductivity). These results are consistent
103 with current knowledge on transfer pathways of nutrients in catchments, with deeper
104 flowpaths for N species than for P species (Strohmer et al. 2020), and explains why
105 landscape features had less influence on NO_3^- than P (Thomas and Abbott 2018). These
106 landscape metrics, however, are relatively rigid, as topological influences (e.g. hydraulic
107 distance to stream, surface flow accumulation, slope), whether considered in the metrics or
108 not, cannot be weighted. In addition, certain topological features vary over a much wider
109 range of values than others, which may obscure other features when no weighting coefficient
110 is included in the landscape metric. For example, flow accumulation (which can have large
111 values) can overshadow the influence of slope (which varies over a narrow range of values)
112 in an index that considers both without weighting them.

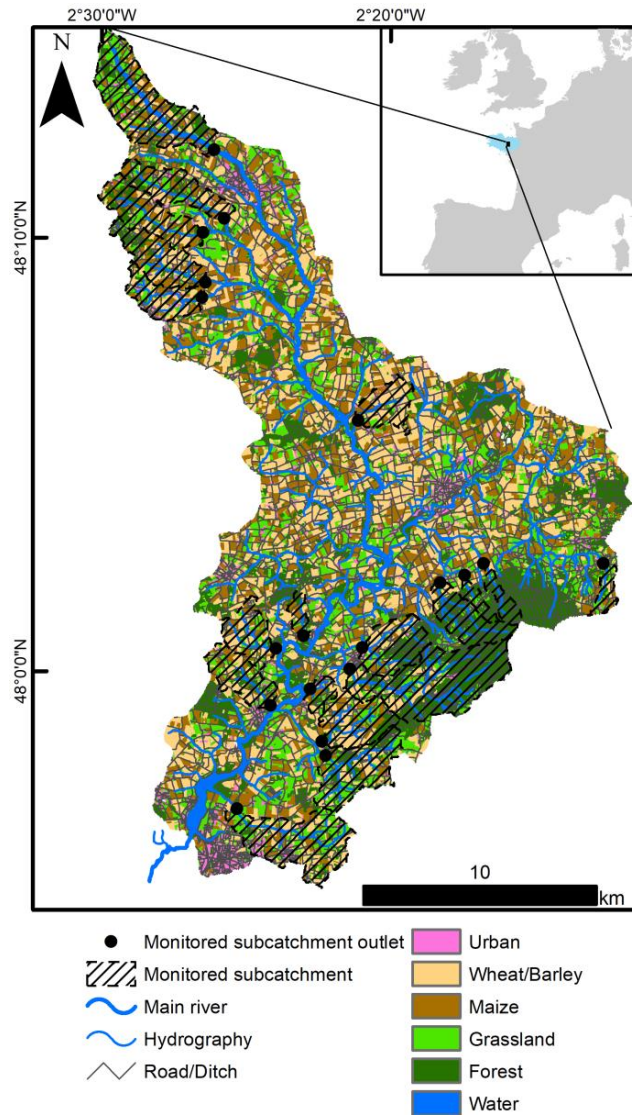
113 The idea that landscape configuration influences N and P loads is the basis for the concept
114 of critical source area (CSA), i.e. the idea that a small percentage of the agricultural area
115 (e.g. <20%) contributes disproportionately to the nutrients transferred to streams (e.g.
116 >80%). CSAs are defined as the intersection of nutrient sources and hydrologically sensitive
117 areas (Gburek and Sharpley 1998; Pionke et al. 2000). Initially defined at the sub-field scale,
118 the concept of CSA was extended to larger scales; entire fields or subcatchments can also
119 be CSAs (Buchanan et al. 2013b; Page et al. 2005; Reaney et al. 2019; Sharpley et al. 2011;
120 Srinivasan and McDowell 2009). Substantial uncertainties remain, however, in their
121 delineation at all scales (Doody et al. 2012). Validation at the sub-field scale can be based on
122 observing erosion marks (Reaney et al. 2019) or tracers (Collins et al. 2012). However,
123 studies that validate CSA delineation based on N or P concentrations in streams and rivers
124 are rare and are based on only a few hydrologically contrasting headwater catchments
125 (McDowell and Srinivasan 2009; Shore et al. 2014; Thomas et al. 2016a) or larger
126 catchments for which land-use composition is already a good predictor (Giri et al. 2018).

127 To address the limits of “expert-based” delineation of CSAs and the rigidity of spatially
128 explicit landscape metrics, we developed a stochastic, data-driven approach based on 30
129 synoptic samplings of 19 agricultural headwater catchments to answer two questions: i) Does
130 landscape configuration influence N and P exports?; and ii) Does the location of CSAs
131 depend on the hydrological distance of the nutrient source to the hydrological network, their
132 overlap with flow accumulation areas, or both? We then investigated why some relationships
133 between water quality parameters and landscape composition metrics break down below a
134 certain catchment size. For this, we studied how the relationship between landscape
135 composition and configuration varied as a function of catchment size in 500 randomly
136 selected subcatchments in the study area.

137 **2. Methods**

138 **2.1. Study site**

139 The Yvel catchment is a 375 km² agricultural catchment of Strahler order 5 in Brittany,
140 western France (Fig. 1). The Yvel River is the main tributary of a three million m³ water
141 reservoir that has been subjected to cyanobacteria blooms since the 1970s (ODEM 2012),
142 for which N and P are deemed responsible (Shatwell and Köhler 2018).



143

144 **Fig. 1** Monitored subcatchments, land use in 2018, hydrography and roads in the Yvel
 145 catchment. The inset shows the location of Brittany, France, in western Europe

146 The climate is temperate oceanic with mean annual precipitation (1998-2017) of 777 mm
 147 (sd=132 mm), mean annual temperature of 11.7°C (sd=0.5°C), and mean annual runoff of
 148 254 mm (sd=143 mm). The Yvel River's discharge is monitored for 300 km² of its 375 km²,
 149 and monthly mean discharge varies from 157 L.s⁻¹ in August to 5,600 L.s⁻¹ in February. The
 150 hydrology is controlled by the dynamics of the shallow groundwater within unconsolidated
 151 weathered material that caps impervious schist bedrock (Casquin et al. 2020). The land use
 152 is dominated by arable fields (maize and winter cereals), which cover 54% of the catchment
 153 (Fig. 1). Grasslands (21%, mainly leys in rotation), forests (18%), and urban areas (6%)
 154 comprise the rest of the catchment area. Hedgerow density is 71 m.ha⁻¹. Soils in the
 155 catchment are generally shallow (<100 cm), are well-drained in the upland part of the
 156 hillslope, and are often hydromorphic in valley bottoms. The elevation varies from 33-297 m
 157 above sea level. The centre of the catchment is the flattest area (most slopes <5%), with long

158 and regular hillslopes. In the north and south, the relief is more rugged, with shorter and
159 steeper slopes. The southeast portion of the watershed of the catchment is forested and has
160 the steepest slopes (5-15%).

161 2.2. Subcatchment monitoring

162 The monitoring strategy consisted of repeated synoptic sampling of 19 subcatchments (Fig.
163 1). The 19 subcatchments were selected based on Strahler order (1-2), size (0.8-12.6 km²,
164 mean=5.1 km²), absence of a wastewater treatment plant, and accessibility. Their
165 percentage of agricultural land-use ranged from 17-94% (mean=74%), with mean slopes
166 ranging from 2.7-6.6% (mean=4.9%). Together, these 19 subcatchments covered 28% of the
167 Yvel catchment's area. All 19 monitoring points were sampled approximately every two
168 weeks from April 2018 to July 2019 (30 dates in total). Samples were filtered *in situ*
169 immediately after sampling with cellulose acetate filters of 0.20 µm pore size for nitrate (NO₃⁻)
170 analysis. All filters were rinsed in the laboratory with 20 ml of deionised water before use. An
171 unfiltered water sample was also collected to analyse TP. The samples were transported to
172 the laboratory in a cool box and then refrigerated at 4°C until analysis. TP was analysed
173 within 48 h of sampling, while NO₃⁻ was analysed within one week. TP was determined
174 colorimetrically via reaction with ammonium molybdate (Murphy and Riley 1962) after
175 digesting the samples in acidic potassium persulfate. The precision of TP measurements
176 was ±13 µg.L⁻¹, while that of NO₃⁻ concentrations, analysed by ionic chromatography
177 (Dionex, DX120), was ±4%. Hydrochemical data are available at
178 <https://www.hydroshare.org/resource/7c7d7f6dd1f14450883ae1c243c3c28f/> (Dupas 2020).

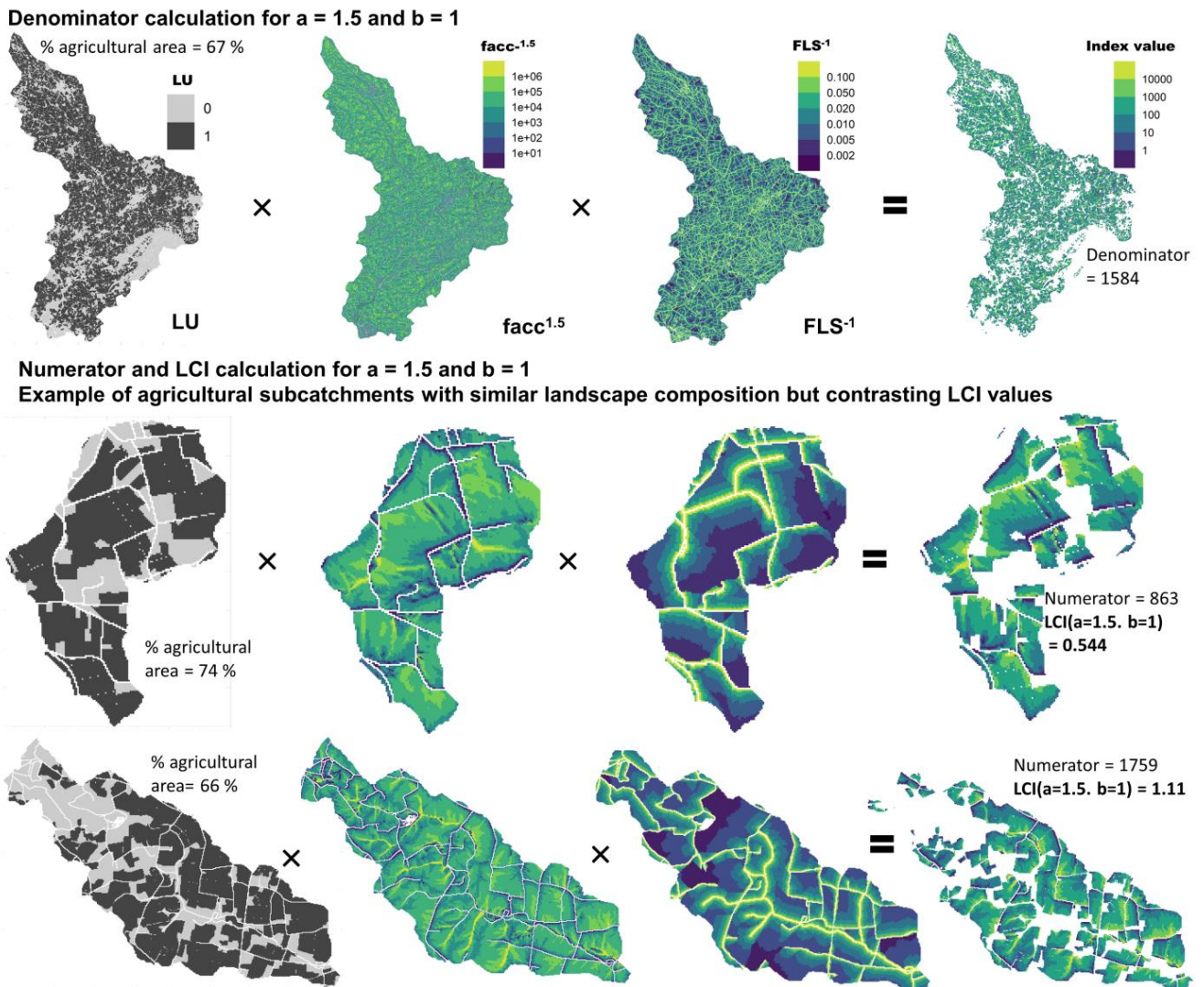
179 2.3. Landscape configuration index

180 We developed the landscape configuration index (LCI) (Eq. 1) to test the hypothesis that the
181 hydrological proximity of agricultural areas to watercourses and their overlap with flow
182 accumulation areas influence nutrient exports at the headwater catchment scale. For each
183 monitored subcatchment, the index was calculated as follows:

$$184 \quad LCI(a,b) = \frac{\frac{1}{n} \times \sum_{i=1}^n LU_i \times FAcc_i^a \times FLS_i^{-b}}{\frac{1}{N} \times \sum_{i=1}^N LU_i \times FAcc_i^a \times FLS_i^{-b}} \quad (Eq. 1)$$

185 where a and b are calibrated parameters, i=1..n are the pixels of a subcatchment; i=1..N are
186 the pixels of the entire catchment, FAcc_i is the flow accumulation on pixel i (m²), FLS_i (m) is
187 the distance along the surface flow line from pixel i to the stream/ditch, and LU_i equals 1 if
188 pixel i is a source of nutrients (i.e. an agricultural area), otherwise 0.

189 Fig. 2 details the calculations for two subcatchments with similar land-use composition, but
 190 whose LCI numerator varies by a factor of 2 when parameters (a, b) = (1.5, 1).



191
 192 **Fig. 2** Steps used to calculate the index given a=1.5 and b=1.0 for 2 of the 19 monitored
 193 subcatchments. Note the logarithmic colour scale

194 The denominator is a normalisation factor that corresponds to the mean value of the
 195 numerator for the entire Yvel catchment. When LCI > 1, a subcatchment's nutrient sources
 196 are located predominantly in flow accumulation zones and/or near streams. For large values
 197 of a or b, the LCI assigns high weights to a small percentage of the area. We restricted the
 198 ranges of parameters a and b so that a few pixels with the highest Facc and lowest FLS
 199 would not control the values of LCI. We explored the parameter space for pairs of (a, b) for
 200 which the highest 5% of cumulative LCI in the landscape did not exceed 95% of the overall
 201 cumulative LCI. When b=0, we varied a within the range 0-2, whereas when a=0, we varied b
 202 within the range 0-4.

203 By construction, when a and b equal 0 for a subcatchment, its LCI equals its percentage of
204 agricultural land-use (i.e. landscape composition) divided by the percentage of agricultural
205 land-use in the entire Yvel catchment. For other values of a and b, the LCI indicates the
206 landscape configuration. High values of parameter a increase the weight of pixels in flow
207 accumulation zones in the LCI, while high values of parameter b increase the weight of pixels
208 in near-stream zones.

209 **2.4. Spatial data sources and pre-processing**

210 **2.4.1. Hydrographic network**

211 The hydrographic network consisted of both the “natural” stream network and the ditch
212 network because 1) it is often difficult to distinguish a ditch from a rectified stream and 2) our
213 synoptic sampling verified that most of the ditches were deep (> 1 m) and flowed in winter
214 (high-flow season). Ditch networks are a shortcut between agricultural areas and the
215 "natural" river network (Ahiablame et al. 2011; Buchanan et al. 2013a). Moreover, evidence
216 suggests that ditches act as 1st order streams when considering nutrient spiralling and can
217 retain and remobilise N and P (Dunne et al. 2007; Hill and Robinson 2012; Smith 2009).
218 Thus, we considered them part of the hydrographic network. We used the road network as a
219 proxy for the ditch network, as we observed that ditches bordered all roads in the study area.
220 Streams (permanent and intermittent) and roads were extracted from vector data (accuracy
221 of ca. 1.5 m) provided by the *Institut National de l'Information Géographique et Forestière*
222 (IGN) at 1:25,000 scale. We converted these vector data to raster format and aligned them
223 with the Digital Elevation Model (DEM) for later analysis.

224 **2.4.2. Digital Elevation Model**

225 The DEM, with a native resolution of 5 m (IGN 2018), was resampled to 10 m using cubic
226 splines in ArcGIS 10.6. Filling was used to remove the depressions on hillslopes (Planchon
227 and Darboux 2002), and the value "NA" was assigned to the pixels corresponding to roads
228 and streams. We calculated flow accumulation and the hydrological distances to streams (i.e.
229 following the surface flow paths) using the multiple flow direction algorithm (Qin et al. 2007).
230 We chose this algorithm for its ability to generate realistic flow accumulation maps, unlike the
231 D8 algorithm (O'Callaghan and Mark 1984).

232 **2.4.3. Nutrient sources**

233 We extracted the agricultural area from the national Land Parcel Identification System
234 (Levavasseur et al. 2016). The data are provided as a vector dataset at the 1:5000 scale for
235 each year since 2010 and contain field boundaries and a code that identifies the crop type.
236 We used the 2018 dataset and verified the spatial accuracy of the agricultural area
237 boundaries based on 50 cm orthophotos and the hydrography. We then rasterized this

238 dataset aligned with the DEM. We assigned the value 1 to agricultural areas and 0 to non-
239 agricultural areas. We included riparian buffer strips in agricultural areas because they are
240 recent and have been fertilised for years, and are a well-documented legacy source of
241 nutrients in headwater catchments (Gu et al. 2018).

242 **2.5. Optimisation of (a, b) parameters and interpretation**

243 For each pair (a, b) within the previously defined domain of exploration, we calculated
244 Spearman's rank correlation (R_s) between the LCI and median NO_3^- and TP concentrations
245 of each subcatchment (Figure 3). Since several of the subcatchments were intermittent, we
246 calculated the median concentrations for the 22 dates (out of 30) when at least 17 of the 19
247 subcatchments were flowing, so as not to bias calculation of the median. We focused on
248 ranks rather than concentrations because several studies have shown that concentration
249 estimates had high uncertainty when calculated with low-frequency data (e.g. Cassidy and
250 Jordan 2011; Moatar et al. 2020), while the ranks of subcatchments could be predicted with
251 high degree of confidence, as they are stable across flow conditions (Abbott et al. 2018;
252 Dupas et al. 2019, Gu et al., 2020 - in review). The result interpretation was twofold:

- 253 - The spatial configuration of sources was considered to have an effect if R_s for at least
254 one pair (a, b) \neq (0, 0) was significantly higher than that for (a, b) = (0, 0).
- 255 - Optimal values of a and b were examined to assess the relative importance of
256 hydrological distance to streams and flow accumulation on hillslopes.

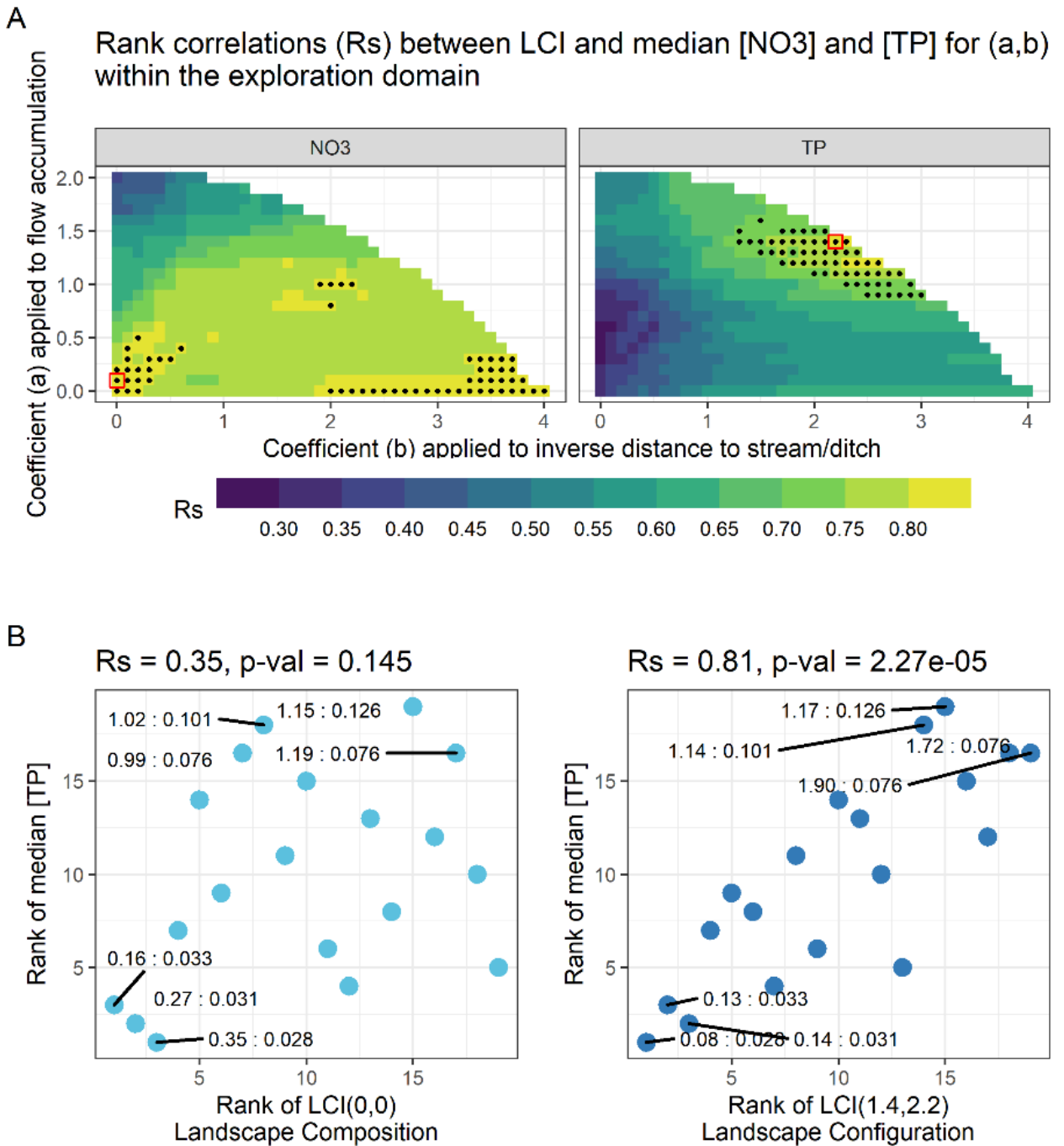
257 We then calculated the ratio of LCI with optimal (a, b) to LCI with (a, b) = (0, 0) for 500
258 randomly selected subcatchments in the Yvel catchment, and analysed how the relationship
259 between these two metrics evolved as catchment size increased.

260

261 **3. Results**

262 **3.1. Comparison of land-use composition and configuration metrics as predictors of** 263 **NO_3^- and TP concentrations in headwater catchments**

264 Landscape composition (i.e. LCI with (a, b) = (0, 0)) predicted median NO_3^- concentrations
265 well ($R_s=0.84$, $p<0.001$) but not those of TP ($R_s=0.33$, $p=0.18$). Varying (a, b) did not
266 substantially improved the prediction of median NO_3^- concentration rank; thus, landscape
267 composition predicted NO_3^- exports well at the headwater catchment scale, and considering
268 landscape configuration did not improve the prediction (Fig. 3A). For NO_3^- , this result refutes
269 our first hypothesis that the spatial configuration of nutrient sources influences nutrient
270 exports.

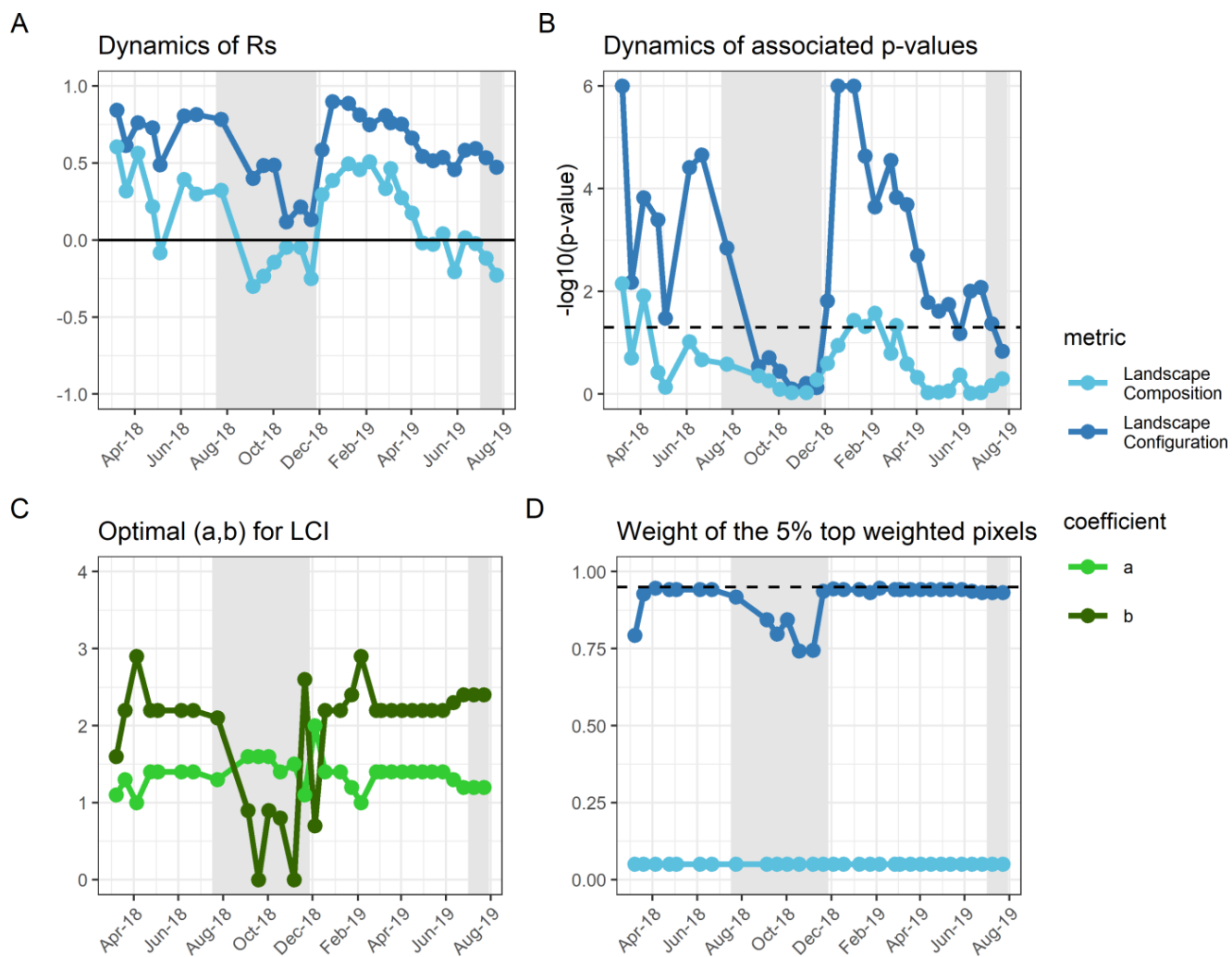


272

273 **Fig. 3** A) Optimisation plan for parameters a and b used to predict median concentration of
 274 nitrate (NO₃⁻) and total phosphorus (TP) in 19 headwater catchments. Rank correlations (R_s)
 275 not shown (blank) indicate (a, b) outside of the exploration domain. For each optimisation
 276 plan, black dots indicate 10% highest R_s , red square the best correlation B) R_s between
 277 median [TP] and 1) landscape composition and 2) optimised landscape configuration index
 278 (LCI). Annotations show LCI:median [TP]

279 The correlation between the percentage of agricultural land and median TP concentrations
280 ((a, b) = (0, 0)) was 0.35 ($p=0.145$) (Fig. 3B). This correlation improved as weights increased
281 for sources near streams (R_s increased as b increased) and for sources that overlapped
282 surface flow accumulation (R_s increased as a increased) (Fig. 3A). This confirms our first
283 hypothesis that landscape configuration influences P exports, and agricultural areas near
284 watercourses and that overlap surface flow accumulation areas result in larger P exports.
285 The optimum correlation was obtained for $a=1.4$ and $b=2.2$ ($R_s=0.81$, $p<0.001$), at the limit of
286 our field of exploration, which means that 5% of the area determined nearly 95% of the
287 index.

288 Using the same parameter exploration scheme, we optimised (a, b) for TP concentrations on
289 each sampling date (Fig. 4C). The LCI with (a, b) \neq (0, 0) always predicted TP concentrations
290 better than landscape composition (Fig. 4A). The correlation between the optimised LCI and
291 TP concentrations differed significantly ($p<0.05$) from 0 for all but one sampling date, except
292 during the low-flow periods (Fig. 4B).



293

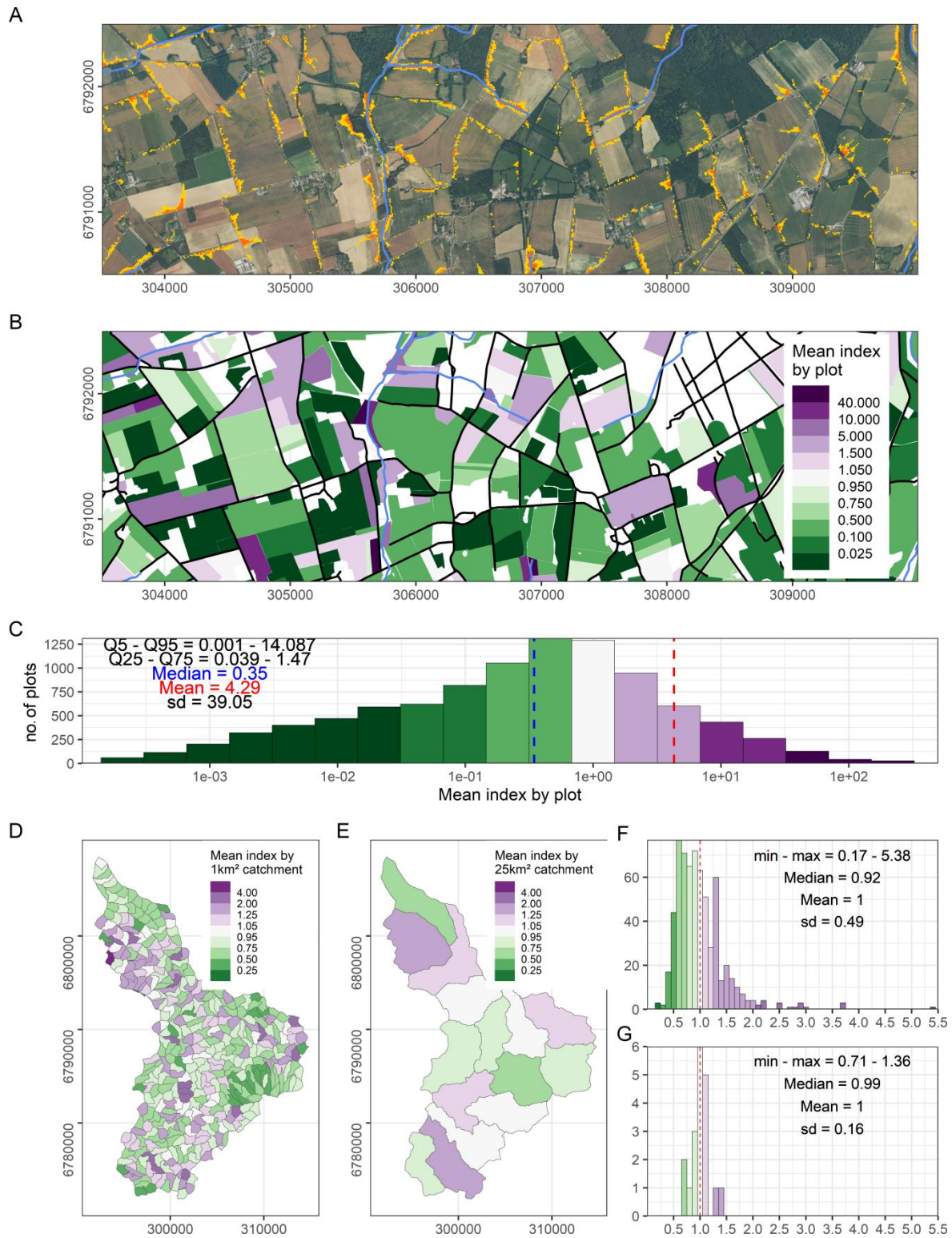
294 **Fig. 4** Comparison of landscape composition and optimised landscape configuration index
 295 (LCI) as predictors of TP concentrations, for all sampling dates. (a) Spearman's rank
 296 correlation (Rs), (b) $-\log_{10}$ -transformed p-values (dashed line indicates $p=0.05$), and (c) for
 297 optimised LCI, parameters a and b applied to flow accumulation and inverse distance to the
 298 stream/ditch (d). Relative weight of the top 5% of weighted pixels. Grey areas represent low-
 299 flow periods, when two or more sampled streams were dry

300 During the low-flow season (Aug-Dec 2018) and beginning of the rewetting season (Jul
 301 2019), TP concentrations in the headwaters were not correlate with the optimised LCI
 302 ($p \gg 0.05$) (Fig. 4B). Note that 13 of the 19 streams were dry at the peak of the low-flow
 303 season (Fig. 1S). Outside the low-flow season, optimal values of parameters a and b were
 304 stable, varying little around a median value of 1.4 and 2.2, respectively. Therefore, the total
 305 weight of the top 5% of weighted area remained close to 95% (Fig. 4D), except during the
 306 low-flow season. The optimised LCI based on median TP concentrations, hereafter referred
 307 to as LCI-TP, appears to be a robust sensitivity index to determine TP CSAs across flow
 308 conditions.

309 **3.2. Spatial aggregations of the LCI-TP**

310 The data-driven optimization of the LCI parameters to predict the ranking of exports at the
311 catchment scale yielded an index that assigned nearly all of the weights (95%) to small areas
312 (5%). Most of these areas (Fig. 5A) were located near streams and ditches, but their width
313 varied. The CSAs often extended ca. 100 m or more into agricultural areas, especially on
314 long and convex hillslopes. Using LCI-TP values at the pixel level, we calculated $\text{LCI-TP}_{\text{field}}$
315 as the mean LCI-TP of each agricultural field. The $\text{LCI-TP}_{\text{field}}$ (Fig. 5B) revealed that most
316 fields were weak sources of P (i.e. $\text{LCI-TP}_{\text{field}} \ll 1$), whereas a few fields were CSAs, as their
317 $\text{LCI-TP}_{\text{field}}$ exceeded 10 and even 40. The associated histogram (Fig. 5C) followed a
318 lognormal distribution, which confirms the high variability at the field scale ($Q5-Q95=0.001-$
319 14.087 , $sd=39.05$). When the LCI-TP was aggregated into 1 km^2 subcatchments ($\text{LCI-TP}_{1\text{km}^2}$)
320 (Fig. 5D), its variability decreased drastically (Fig. 5F, $\text{min-max}=0.17-5.38$, $sd=0.49$). The
321 number of subcatchments with $\text{LCI-TP}_{1\text{km}^2} > 1$ was approximately the same as the number of
322 subcatchments with $\text{LCI-TP}_{1\text{km}^2} < 1$ (Fig. 5F, $\text{median}=0.92$), but the distribution remained
323 lognormal. Following the same pattern, the LCI-TP aggregated into 25 km^2 subcatchments
324 ($\text{LCI-TP}_{25\text{km}^2}$) had even lower variability ($\text{min-max}=0.71-1.36$): at this scale the distribution
325 was symmetrical (apparently normal), and no subcatchment could be considered a CSA. The
326 information at the subfield scale (Fig. 5A) and field scale (Fig. 5B), which is relevant for
327 farmers and catchment managers, was generated for the entire study area ([https://antoine-](https://antoine-csgn.github.io/YV1.html)
328 [csgn.github.io/YV1.html](https://antoine-csgn.github.io/YV1.html)).

329

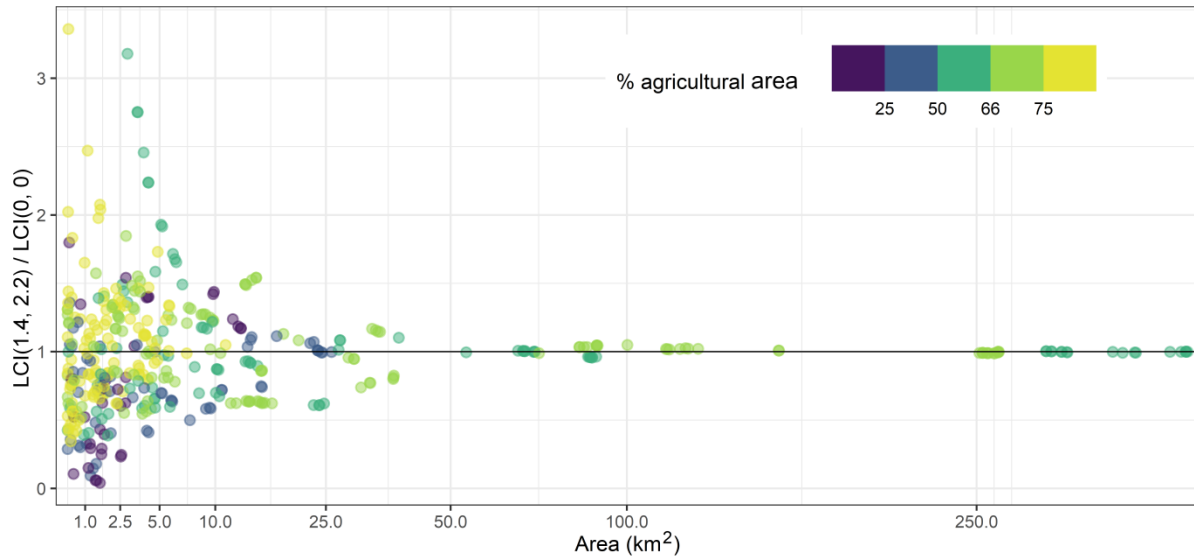


330

331 **Fig. 5** A) Excerpt of the study area, ca. 6 km × 2 km: top 5% (yellow) and 1% (red) of
 332 weighted pixels according to LCI-TP, B) Mean LCI-TP aggregated at the field scale (LCI-TP
 333 *field*), and C) histogram of associated values (note the log scale on the x-axis). D) LCI-TP for 1
 334 km² subcatchments (LCI-TP_{1km²}) and E) 25 km² subcatchments (LCI-TP_{25km²}) and associated
 335 histograms (F and G, respectively)

336 **3.3. Homogenisation of the LCI-TP with increasing catchment size**

337 We delineated 500 subcatchments within the study area based on 500 points randomly
338 generated over the hydrographic network. For each subcatchment, we calculated the LCI-TP
339 (i.e. the LCI for $a=1.4$ and $b=2.2$), the landscape composition index (i.e. the LCI for $a=b=0$),
340 and their ratio, which we examined as a function of subcatchment area (Fig. 6).



341
342 **Fig. 6** Stabilisation of the ratio of the LCI-TP (landscape configuration index (LCI) for $a=1.4$
343 and $b=2.2$) to landscape composition index (LCI for $a=b=0$) as a function of area (square-
344 root-transformed x-axis) of subcatchments in the study area (points)

345 This ratio varied greatly (0.02-3.37) for subcatchments smaller than 10 km², the typical size
346 of 1st or 2nd order stream catchments but varied less (0.55-1.53) for subcatchments of 10-50
347 km². The high variability in this ratio, even for heavily farmed subcatchments, indicates that
348 the landscape can still be optimised at this scale. For catchments larger than 50 km², which
349 corresponds to 4th order rivers in the study area, the ratio converged to 1. The main
350 implication is that for headwater catchments (<50 km²) the correlation between landscape
351 composition and configuration (as defined by the LCI-TP) is weak and non-significant (Fig.
352 2S), while for larger catchments the correlation is strong (R^2 near 1) (Fig. 2S).

353
354 **4. Discussion**

355 **4.1. Spatial configuration of nutrient sources influences P but not N exports**

356 Landscape composition is a strong predictor of median N concentrations, while landscape
357 configuration as defined here, seems to have no influence (Fig. 2A). This result agrees with
358 previous research that indicates a strong correlation between metrics of agricultural N
359 pressure intensity, such as N surplus or N concentrations in streams (Dupas et al. 2015;

360 Kronvang et al. 2005), leaving little space for other controlling factors, such as landscape
361 configuration. However, distributed process-based modelling of N fate in agricultural
362 catchments has shown that landscape configuration can have a of second-order influence on
363 N concentrations (Beaujouan et al. 2002; Casal et al. 2019; McDowell et al. 2014), which
364 may not have been captured in our study. P concentrations are not related to landscape
365 composition (Bol et al. 2018) or even P inputs (Dupas et al. 2015; Frei et al. 2020) in
366 headwater catchments. Multiple factors influence P transfers to streams: soil type and P
367 content (Djodjic et al. 2004), tile-drainage (King et al. 2015a; King et al. 2015b), small ponds
368 and hillside storage reservoirs (Schmadel et al. 2019), in-hillslope depressions (Smith and
369 Livingston 2013), leaks from septic tanks (Withers et al. 2011) and livestock buildings, bank
370 erosion (Kronvang et al. 2012), and ditch-dredging management (Smith and Pappas 2007).
371 While hedgerows have little influence on N transfers at the catchment scale (Benhamou et al.
372 2013; Thomas and Abbott 2018), it is difficult to quantify their combined effects on P
373 transfers. Without considering these factors, our optimised LCI ranked the headwater
374 catchments reasonably well according to their P exports (Fig. 2B), although the residuals
375 may have been due to these other factors. We demonstrate that the spatial configuration of P
376 sources in the landscape is critical to understand P transfer from land to streams in
377 headwater catchments. The influence of the spatial configuration of nutrient sources on P but
378 not N concentrations in agricultural headwater streams results in high variability in N:P ratios
379 in these ecosystems and has many consequences on stream algae communities (Pringle
380 1990; Stelzer and Lamberti 2001).

381 The observation that landscape spatial organisation influences N and P transfers is not new.
382 Nonetheless, our approach differs from previous studies that used single or multiple
383 regression of several landscape metrics compiled in FRAGSTATS (McGarigal and Marks
384 1995) to predict nutrient exports at the catchment scale. Despite the large body of studies
385 that use these landscape metrics (Bu et al. 2014; Lee et al. 2009; Ouyang et al. 2014; Zhang
386 et al. 2019), no robust cause-effect relationships emerge, and many of them give conflicting
387 results (Wang et al. 2020). Three reasons may explain why these studies, taken together,
388 are not conclusive: i) the many pairwise or multiple regressions between water quality
389 parameters and landscape metrics generate spurious correlations that are not generally
390 discussed, ii) landscape metrics depend greatly on the resolution of GIS input data, and iii)
391 the studies do not consider the topological dimension, which is fundamental for explaining
392 hillslope-to-stream transfers and landscape configuration (Thomas et al. 2016b). Several
393 approaches have integrated this topological control with the hypothesis that nutrient sources
394 near streams (Sliva and Dudley Williams 2001; Yates et al. 2014) and/or that overlap flow
395 accumulation areas (Peterson et al. 2011; Staponites et al. 2019) have a disproportionately

396 higher influence on nutrient exports than other areas. While these approaches still depend on
397 the spatial resolution of GIS data, they are “hypothesis-driven” rather than purely “data-
398 driven”, which decreases the risk of spurious correlations. A longstanding weakness of these
399 approaches, however, is their rigidity due to the lack of calibrated parameters. For example,
400 the HAiFLS index (Peterson et al. 2011) and Flow-A index (Staponites et al. 2019) included
401 both the distance to streams and flow accumulation, but the latter dominates the index value
402 by construction, because it varies more than the former, and neither index allows both factors
403 to be weighted by calibrated coefficients. The stochastic approach in the LCI developed here
404 is more flexible, and we show that both hydrological distance and flow accumulation
405 influence P transfer.

406 **4.2. Spatial variability and temporal stability of critical source areas**

407 The strongest correlation between median NO_3^- concentrations and LCI was for $(a, b) = (0,$
408 $0)$, which shows that each source (agricultural area) contributed the same, regardless of its
409 distance to streams or overlap with flow accumulation areas. This confirms the need to
410 consider the entire catchment to reduce N loads in agricultural catchments. For median P
411 concentrations, the optimal was found for $(a, b) = (1.4, 2.2)$, a value for which 5% of the
412 agricultural area concentrated 95% of the weight assigned by our index at the 10 m-pixel
413 resolution. These values are similar to those found by Thomas et al. (2016a)), who classified
414 1.6-3.4% of the catchment area (during median storm events) and 2.9-8.5% (during upper-
415 quartile events) as prone to P transfer, based on a CSA model that also considered land use
416 and topography as input variables. Summing the LCI-TP at the field scale indicates that the
417 20% of fields at highest risk represent 85% of the total weights, which are the fields on which
418 mitigation measures should be prioritised. The distribution is asymmetrical, with 69% of fields
419 having a mean $\text{LCI-TP}_{\text{field}}$ less than 1. This information at multiple scales can be a tool to
420 maximise ecosystem services at the catchment scale, by reorganising landscapes to
421 decrease P transfer without increasing the percentage of set-aside areas (Doody et al.
422 2016).

423 The shape and location of the sub-field CSAs overlap both the mandatory riparian buffer
424 strips (5 m according to local application of the European Union Nitrate Directive (DREAL
425 2018)) and in-hillslope CSAs based on the Topographical Wetness Index. Because most
426 buffer strips in the study area were installed recently, we included them in P sources as they
427 were enriched in P before conversion (Dodd and Sharpley 2016; Gu et al. 2017; Gu et al.
428 2018; Jarvie et al. 2017; Roberts et al. 2012). The location and shape of the CSAs indicate
429 the need for new shapes of buffer zones, with variable widths and locations along ditches as
430 well. Because buffer strips are critical sources of nutrients at the headwater catchment scale,
431 they require new management practices. Potential solutions include sowing species that can

432 capture more P (William M. Roberts 2019), mowing and exporting the residues each year
433 (Fiorellino et al. 2017), and applying amendments that have high P-sorbing ability (Borno et
434 al. 2018).

435 The temporal stability of the optimal parameters (a, b) for TP during the flow period is
436 consistent with the spatial stability concept (Abbott et al. 2018), but is contrary to the concept
437 of variable source areas (Collick et al. 2015; Dahlke et al. 2012). Our interpretation of this
438 temporal stability is that even though the soil-to-stream connectivity varied temporally during
439 the sampling period, the TP concentrations observed may reflect the remobilisation of
440 sediments transferred during rare erosion events. Observing storm events that result in
441 surface transfers requires frequent observations (Cassidy and Jordan 2011). During the low-
442 flow dates, the correlations with LCI-TP were not significant: different sources and sinks likely
443 dominate the influence of agricultural areas and their configuration, which predominates
444 during the flow period. Leaks from septic tanks or animal buildings and desorption from
445 sediments are the most likely sources of P during this ecologically sensitive period, while
446 hydrological disconnection, uptake, and sedimentation can be sinks of P (Lannergård et al.
447 2020; Sandstrom et al. 2020).

448 **4.3. Landscape homogenisation with increasing catchment size: consequences for** 449 **management, monitoring, and modelling**

450 When means were aggregated into 1 km² subcatchments, which is the typical size of a 1st
451 order catchment in the study area, the LCI-TP showed high spatial variability (Fig. 5D, 5F).
452 As mentioned, this variability decreased as the aggregation size increased (Fig. 5E, 5G), and
453 landscape composition was weakly correlated with landscape configuration for catchments
454 smaller than 50 km² but strongly correlated for those larger than 50 km² (Fig. 6, Fig. 2S).
455 These results explain, at least partially, why relationships between water quality parameters
456 and landscape composition metrics break down below a certain catchment size (Bol et al.
457 2018). They also shed light on the long-standing difficulty in scaling from nutrient export
458 models at the field scale to those at the catchment scale: the spatial configuration of the
459 nutrient sources can be critical.

460 The variability in the LCI-TP in small subcatchments and its homogenisation in larger
461 catchments is the expression of a degree of unstructured heterogeneity (i.e. randomness) of
462 the spatial configuration of agricultural areas (Musolff et al. 2017). Some almost entirely
463 agricultural subcatchments have a TP sensitivity index less than 1, while some mixed-land-
464 use catchments have a TP sensitivity index greater than 1. The latter provides the
465 opportunity to introduce structured heterogeneity, i.e. to reorganise agricultural activities
466 spatially to reduce P transfers to streams (Musolff et al. 2017).

467 These results have implications for both modelling and monitoring. Semi-distributed models
468 that simulate P exports from an entire catchment with simulation units smaller than 50 km²
469 should include a coefficient to represent the spatial configuration of the agricultural areas. A
470 distributed model should consider ditches (as they are the entry point of P into the
471 hydrographic network) and, especially, the spatial variability in P sources. When monitoring
472 subcatchments that are smaller than the homogenisation threshold and have similar
473 agricultural land-use composition, the observed differences in P loads cannot be related
474 directly to the agricultural practices or soil properties. The spatial configuration of agricultural
475 areas appears to exert a major control on median P concentrations. This is particularly
476 important for targeting measures to the most cost-effective fields (Doody et al. 2016) and
477 could increase in importance due to the recent development of innovative financial tools to
478 improve water quality, such as payment for ecosystem services schemes, whose obligation
479 to achieve results is increasing (Hejnowicz et al. 2014). Our research provides a data-driven
480 method to identify CSAs and thus the most cost-effective fields on which to implement
481 mitigation measures.

482 The 50 km² landscape homogenisation threshold found in this study is similar to the 18-68
483 km² stream-concentration thresholds found by Abbott et al. (2018) for the Rance and Haut-
484 Couesnon catchments, also located in western France. We assume that the optimised
485 coefficients and homogenisation threshold would vary with the topo-climatic conditions and
486 agricultural landscape characteristics. More research is needed to confirm this connection
487 between landscape configuration and P loads in different environmental settings. Given the
488 high heterogeneity of landscape configuration, this research calls for a spatially dense
489 monitoring network of headwater catchments. We also recommend using the LCI to
490 investigate the influence of landscape configuration on other water contaminants that are
491 transferred mainly by surface flow paths, such as pesticides and faecal bacteria.

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496 **Conflicts of interest/Competing interests (include appropriate disclosures)**

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

499 **Ethics approval (include appropriate approvals or waivers)**

500 Not applicable

501 **Consent to participate (include appropriate statements)**

502 Not applicable

503 **Consent for publication (include appropriate statements)**

504 All co-authors – A. Casquin, R. Dupas, S. Gu, E. Couic, G. Gruau and P. Durand – have
505 approved the manuscript and agree with its submission to Landscape Ecology

506 **Availability of data and material (data transparency)**

507 This study is based on a recently published dataset entitled *Repeated synoptic sampling for*
508 *water chemistry monitoring in the Yvel catchment, northwestern France* available at
509 <https://www.hydroshare.org/resource/7c7d7f6dd1f14450883ae1c243c3c28f/> (Dupas *et al.*
510 2020). It contains the hydrochemical data and the geographic information of the study site.

511 **Code availability (software application or custom code)**

512 The code (R script) will be made available along with the article upon acceptance on a github
513 repository.

514 **Authors' contributions**

515 AC: Conceptualization, Methodology, Formal analysis, Visualization, Writing - original draft.
516 RD: Conceptualization, Supervision, Funding acquisition. SG: Sample collection and
517 analysis. EC: Sample collection and analysis. GG: Project administration, Funding
518 acquisition. PD: Supervision. All authors contributed substantially to revisions.

519

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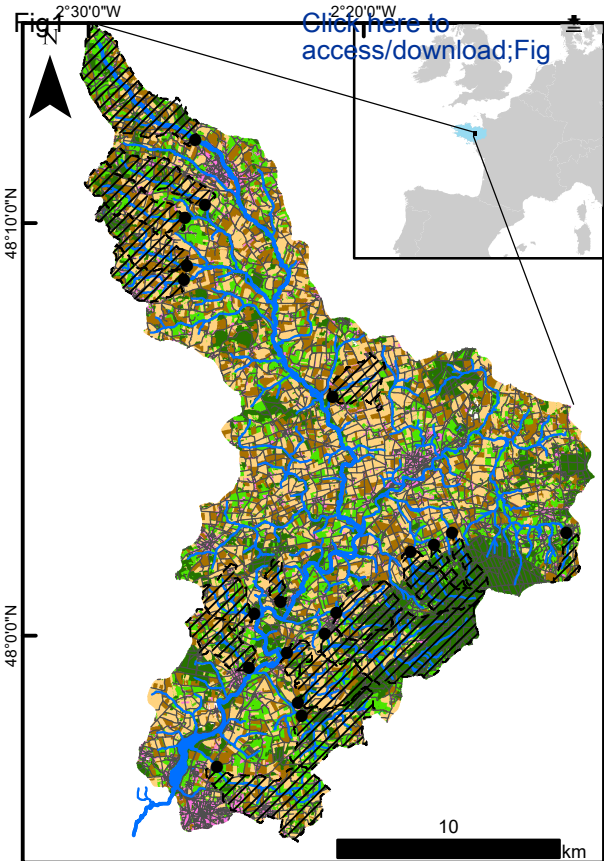
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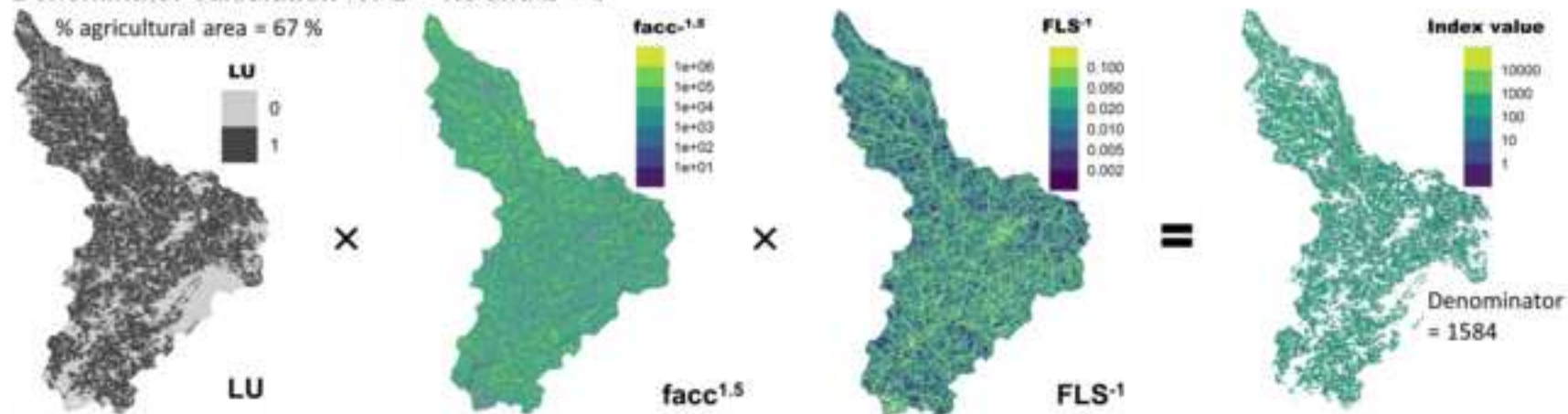
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Denominator calculation for $a = 1.5$ and $b = 1$



Numerator and LCI calculation for $a = 1.5$ and $b = 1$

Example of agricultural subcatchments with similar landscape composition but contrasting LCI values

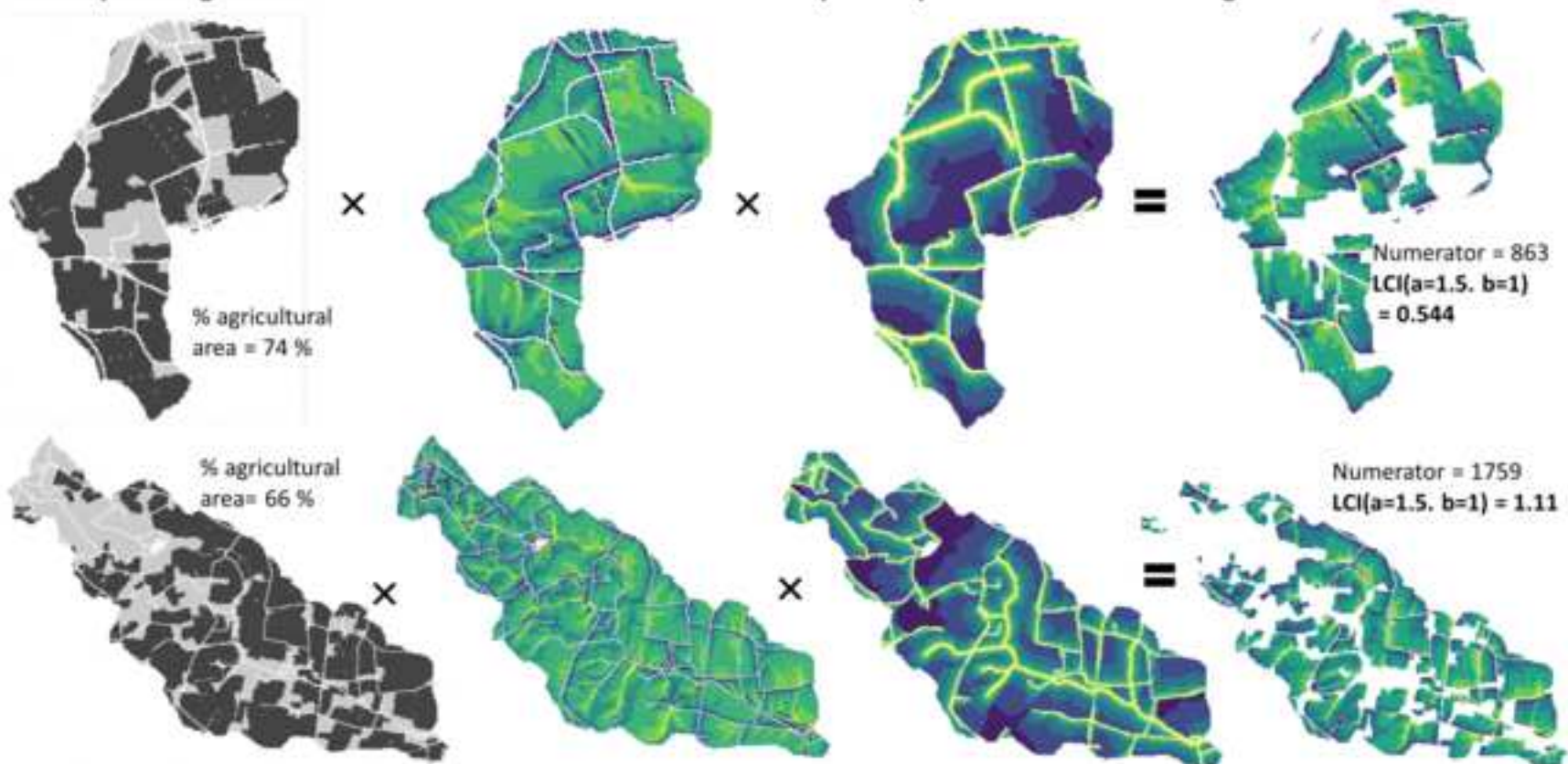


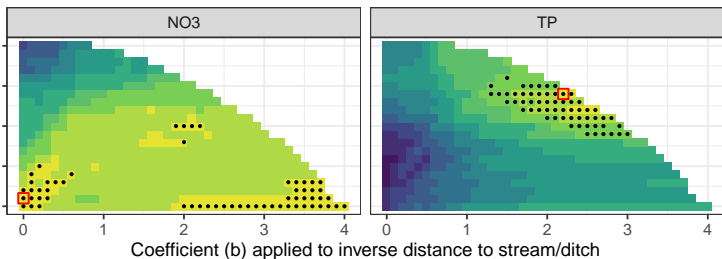
Fig3

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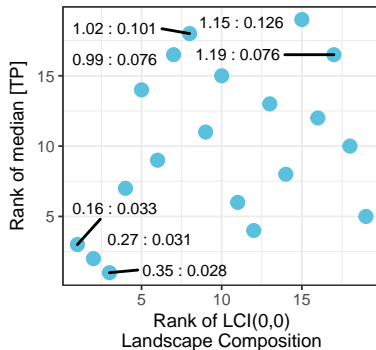
Rank correlations (Rs) between LCI(0,0) and TP (a) and LCI(1.4,2.2) and TP (b) within the exploration domain

Coefficient (a) applied to flow accumulation



B

Rs = 0.35, p-val = 0.145



C

Rs = 0.81, p-val = 2.27e-05

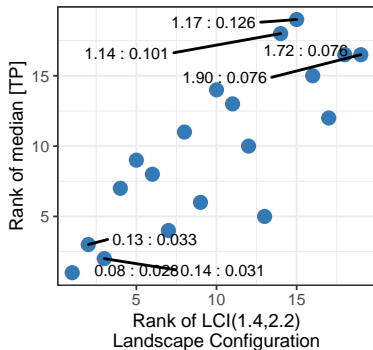
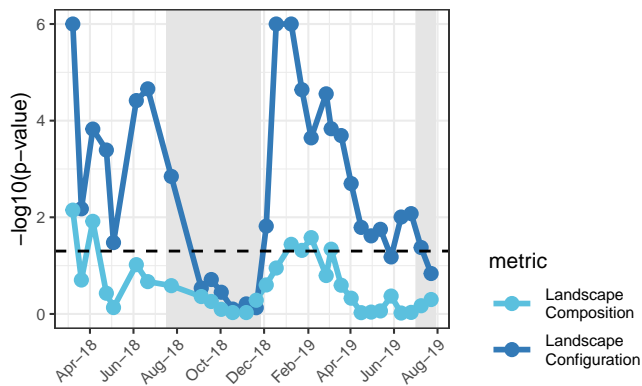


Fig4

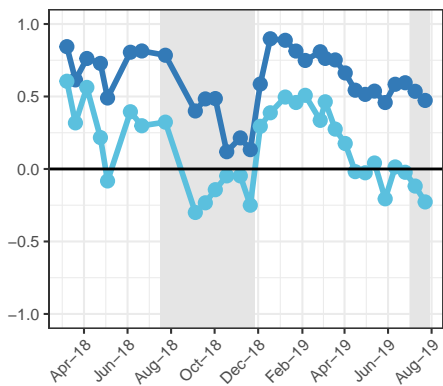
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B

Dynamics of associated p-values

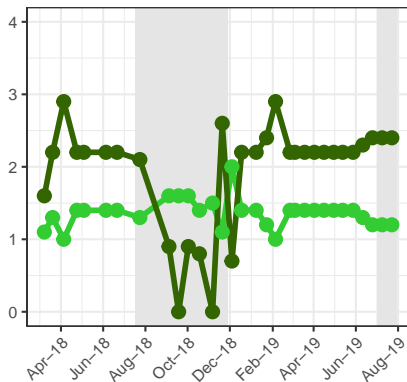


Dynamics of Rs



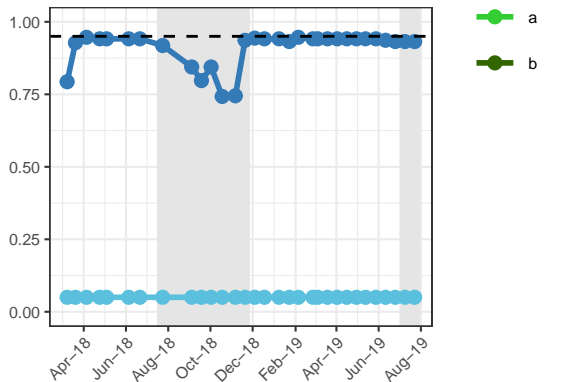
C

Optimal (a,b) for LCI

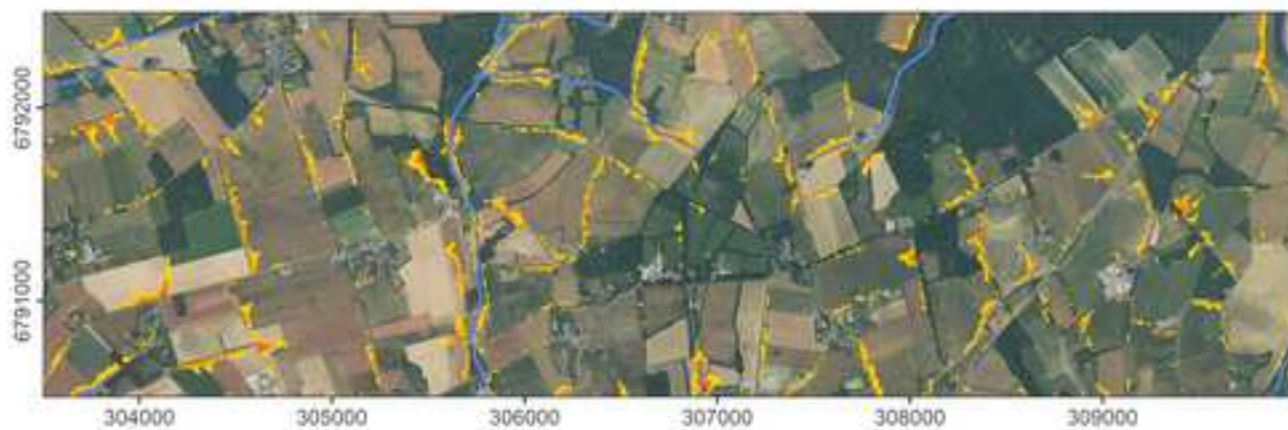


D

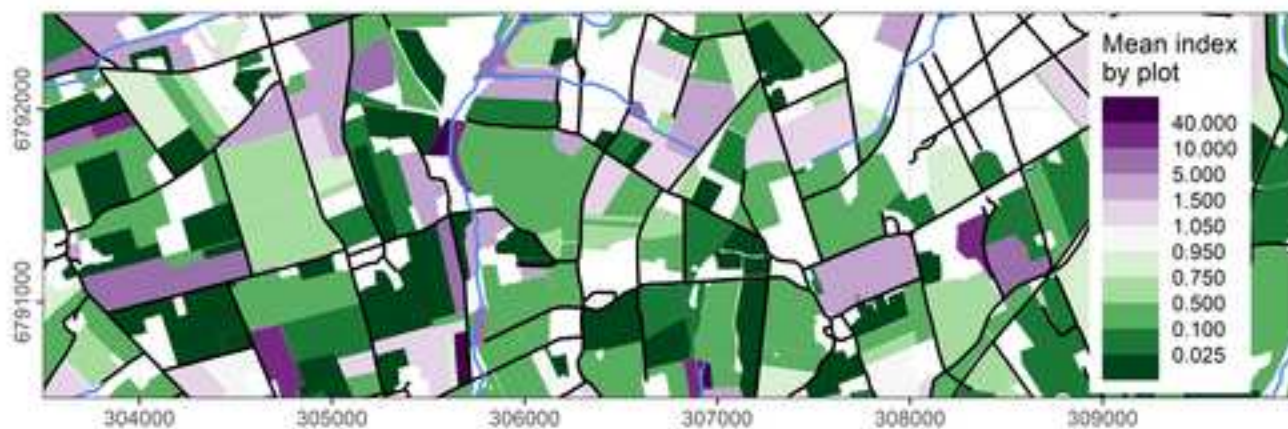
Weight of the 5% top weighted pixels



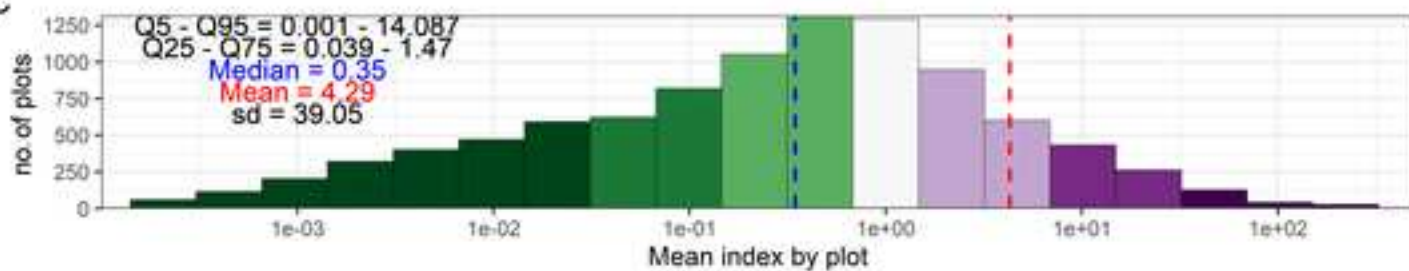
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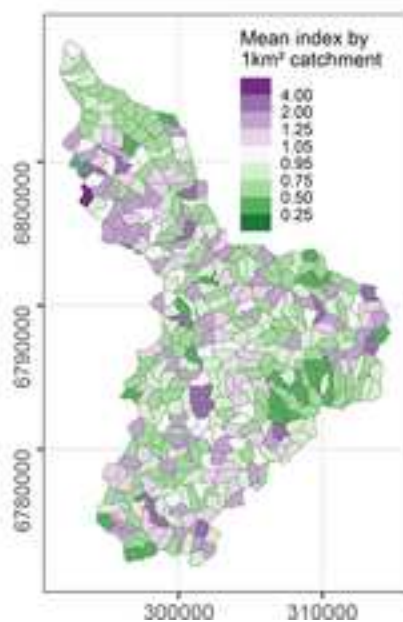
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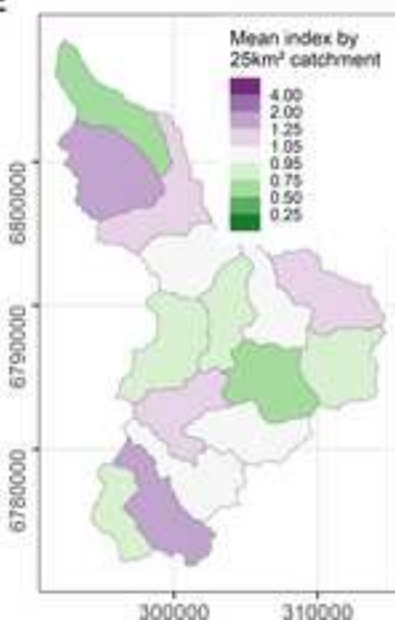
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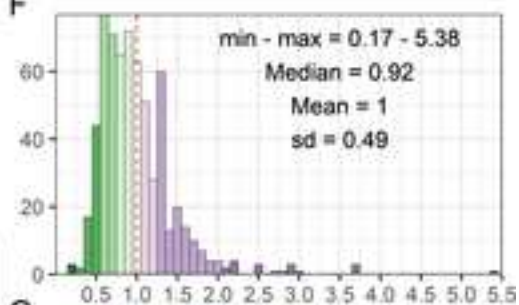
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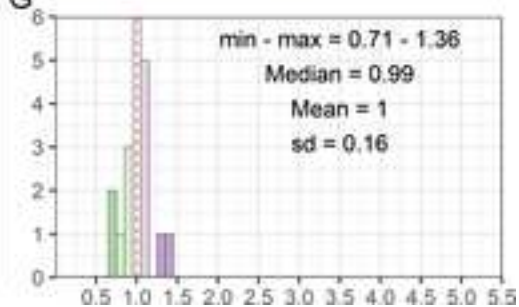
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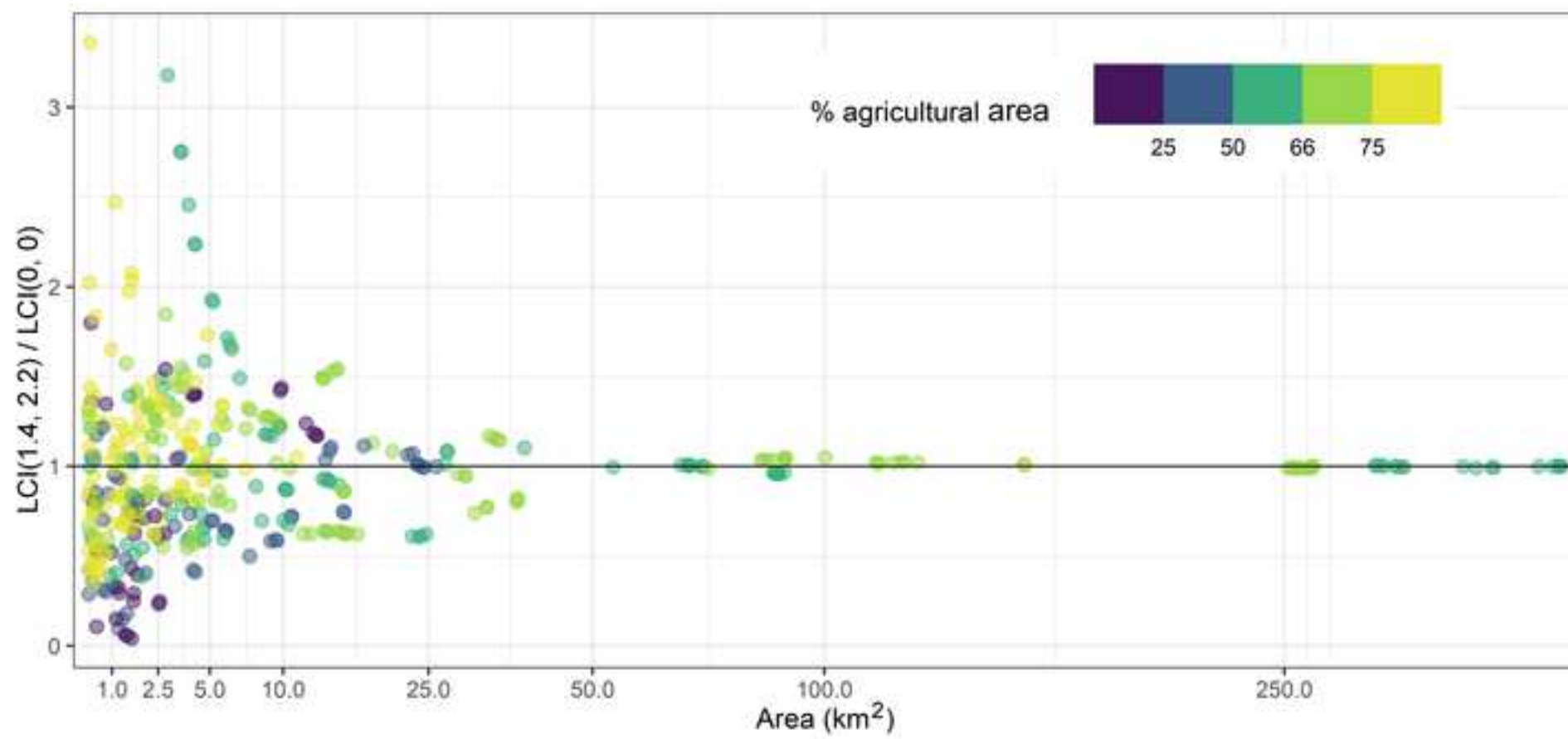


F



G





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Antoine Casquin
INRAE - UMR SAS / Institut Agro
65 rue de St Brieuc
CS 84215
35042 Rennes Cedex
France
Tel: +33223485142
E-mail: antoine.casquin@inrae.fr

Subject: Submission of a manuscript for evaluation and publication in *Landscape Ecology*

Dear Editors,

Enclosed is our manuscript, “The influence of landscape spatial configuration on nitrogen and phosphorus exports in agricultural catchments” by Antoine Casquin et al., which we would like to submit for publication as a Research paper in *Landscape Ecology*.

This manuscript has not been previously published, in whole or in part, and is not under consideration by another journal. All co-authors – A. Casquin, R. Dupas, S. Gu, E. Couic, G. Gruau and P. Durand – have approved the manuscript and agree with its submission to *Landscape Ecology*. We had the English proofread by a professional proofreader (“Editor du Jour”; see enclosed English editing certificate).

The spatial variability in nutrient transfer from land to stream in seemingly similar headwater catchments is currently not fully understood, and upscaling results from the plot scale to the catchment scale remains difficult. Our study indicates that including the influence of the spatial configuration of nutrient sources in relation to the hydrographic network is crucial to explore this issue.

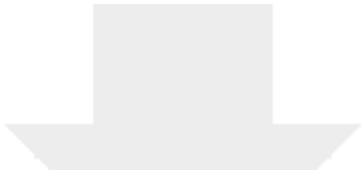
We assessed the influence of landscape spatial arrangement on nitrate (NO_3^-) and total phosphorus (TP) exports at the catchment scale. We developed a novel landscape configuration index that we optimised based on NO_3^- and TP concentrations in 19 headwater streams. Results show that the spatial configuration of nutrient sources and sinks in headwaters has a large influence on TP transfers but not NO_3^- transfers. We also reveal that the spatial configuration of sources is heterogeneous at small scales ($<1 \text{ km}^2$) but homogeneous at larger scales ($> 50 \text{ km}^2$), and we discuss the implications of these findings.

This study is based on a recently published dataset entitled *Repeated synoptic sampling for water chemistry monitoring in the Yvel catchment, northwestern France*. We plan to make the code (R script) available along with the article upon acceptance.

Thank you for considering our manuscript.

Sincerely,

Antoine Casquin



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