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To link to this article DOI: http://dx.doi.org/10.1016/j.scitotenv.2016.09.181

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False Positive and False Negative Errors in the Design and Implementation of Agri-environmental Policies: A Case Study on Water Quality and Agricultural Nutrients

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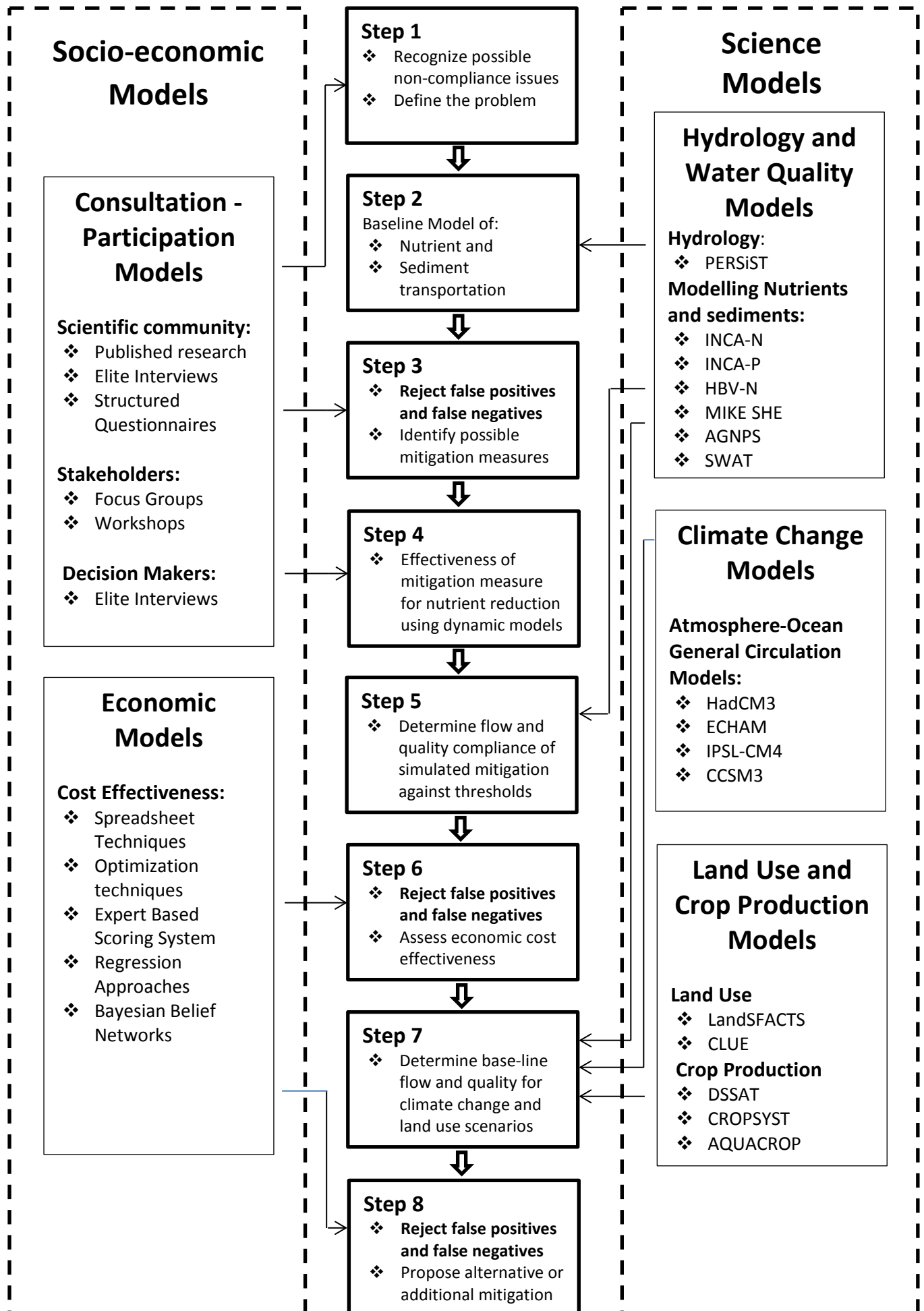
Abstract

When designing and implementing agri-environmental policies to reduce nutrient loss, action programmes may falsely address areas where the nutrient issue from agricultural activity is not currently important and is not likely to become so in the future (a false positive), or may fail to address areas where the agricultural nutrient issue is currently important or may likely become so in the future (a false negative). Based on a case study of the Louros watershed in Greece, this work identifies database and modelling sources of false positives and negatives and proposes a decision making process aimed at minimizing the possibility of committing such errors. The baseline is well simulated and shows that the Louro's watershed falls behind a Good Environmental Status, at least marginally. Simulated mitigation measures show that the river's status can be upgraded to "Good", at least as concerns nitrates and ammonium. Simulated climate change does not seem to exert an important positive or negative effect. Land use changes forecasting considerably less cultivated area have a significant effect on Total Phosphorous but not on nitrates or ammonium concentrations. The non-linearity between nutrient disposition (inputs) and nutrient concentration in downstream water bodies (output) and the many factors that affect the nutrient disposition-transportation-concentration chain, highlights the importance of simulating the effects of mitigation actions and of future climate and land use changes before adopting and establishing agri-environmental measures.

A Decision Making Process for Rejecting False Positives and False Negatives in the Design and Implementation of Agri-environmental Policies

Socio-economic Models

Consultation - Participation Models

Scientific community:
- Published research
- Elite Interviews
- Structured Questionnaires

Stakeholders:
- Focus Groups
- Workshops

Decision Makers:
- Elite Interviews

Economic Models

Cost Effectiveness:
- Spreadsheet Techniques
- Optimization techniques
- Expert Based Scoring System
- Regression Approaches
- Bayesian Belief Networks

Science Models

Hydrology and Water Quality Models

Hydrology:
- PERSiST

Modelling Nutrients and sediments:
- INCA-N
- INCA-P
- HBV-N
- MIKE SHE
- AGNPS
- SWAT

Climate Change Models

Atmosphere-Ocean General Circulation Models:
- HadCM3
- ECHAM
- IPSL-CM4
- CCSM3

Land Use and Crop Production Models

Land Use
- LandSFACTS
- CLUE

Crop Production
- DSSAT
- CROPSYST
- AQUACROP

Step 1
- Recognize possible non-compliance issues
- Define the problem

Step 2
Baseline Model of:
- Nutrient and
- Sediment transportation

Step 3
- **Reject false positives and false negatives**
- Identify possible mitigation measures

Step 4
- Effectiveness of mitigation measure for nutrient reduction using dynamic models

Step 5
- Determine flow and quality compliance of simulated mitigation against thresholds

Step 6
- **Reject false positives and false negatives**
- Assess economic cost effectiveness

Step 7
- Determine base-line flow and quality for climate change and land use scenarios

Step 8
- **Reject false positives and false negatives**
- Propose alternative or additional mitigation

Significant amount of public funds are devoted to EU's agri-environment policy and commitments undertaken by farmers are long term.

It is important to assure policy decision makers that such funds are directed to the areas in need and in a cost effective way

False positive decisions emerge when agricultural activity is falsely acknowledged as the major nutrient supplier or when mitigation measures are falsely assumed to uphold nutrient supply

We advocate a decision making process integrating science and social science models to protect policy design from committing false positives or false negatives

The Louros watershed in Greece is used as a case-study for examining the economic loss under a false positive decision

Climate and land use change can alter the effects of agriculture on water bodies in the future and policy should be prepared to confront this evolution

False Positive and False Negative Errors in the Design and Implementation of Agri-environmental Policies: A Case Study on Water Quality and Agricultural Nutrients

1. Introduction

Mineral fertilizers and livestock manures are the main sources of nutrients which, very often, are out of balance with land availability and in excess of crop needs. This imbalance creates a surplus of nutrients, some of which is lost to water, mainly as nitrates and phosphates, and air mainly as ammonia and nitrogen oxides (MacDonald et al., 2011; Fowler et al., 2015). As a result, eutrophication due to nutrient emission from agriculture and urban and industrial runoff is a major threat to wetland ecosystem health (Verhoeven et al., 2006). In the European Union (EU), agri-environment measures (AEM) constitute one of the main types of policy response for meeting society's demand for environmental outcomes provided by agriculture.

The application of AEM is compulsory at the Member State level, but optional at the farmer level. Consequently, the design of AEM is foreseen to meet public demand for environmental goods under the budgetary constraint of payments to farmers that aim to cover the costs incurred and income forgone as resulting from voluntary environmental commitments. The involvement of farmers is usually medium to long-term with a minimum participation of five years. The agri-environment policy has an embedded "Nitrates" component in its mandatory part, i.e. the Nitrates Directive (EEC, 1991), and implements action programmes for controlling nutrients balance that are voluntary for farmers within the so called Nitrates Vulnerable Zones (NVZs) and through the national and regional Rural Development Programmes (RDPs). The Nitrates Directive is an important building block of the wider European environmental and nature conservation policy as it is directly connected to

1

26   the Water Framework (WFD) and the Habitats and Birds Directives. Over the years,

27   agri-environment policy has emerged as one of the most important elements of the

28   Common Agricultural Policy (CAP) in terms of its budgetary size and the proportion of

29   participating farmers and farmland.

30   The effectiveness of AEMs to enhance biodiversity (Batáry et al., 2015; Kleijn

31   and Sutherland, 2003) and protect aquatic environments from agricultural pollution

32   has been reviewed very extensively, has been questioned and criticized (Grinsven et

33   al., 2016; Buckley et al., 2016; Matzdorf and Lorenz, 2010; Randall et al., 2015). The

34   results are disparate mainly due to the plethora of applied measures, the

35   heterogeneity in the application agroecosystems and their baseline status, the

36   variability in set targets and the way these targets are monitored. Decision making for

37   the adoption and establishment of AEMs targeting the reduction of nutrient

38   concentration in water is implemented, very frequently, without a comprehensive and

39   integrated plan. For example, AEM decision makers may be unable to control for non-

40   agricultural nutrient contributing activities, industrial or municipal, which are beyond

41   their institutional jurisdiction. As a result, AEM decision makers tend to set program

42   targets on inputs (quantities of mineral fertilizers, manure or irrigation water) rather

43   than on downstream chemical water quality or environmental status. Consequently,

44   an AEM can be considered to be very effective because it managed to reduced inputs

45   to the targeted level when, in reality, the AEM had marginal or no effect in reducing

46   nutrient loads downstream.

47   In decision-making, a false positive, known in statistics as Type I error, refers

48   to the situation where the presence of a condition is assumed when in reality there is

49   not such a situation. A false negative, known in statistics as Type II error, refers to the

50 situation where no presence of a condition is assumed when in reality there is one. As

51 such, the words "positive" and "negative" correspond to the answers "yes" or "no" to

52 the question "is upstream agricultural activity responsible for downstream

53 pollution?". In this sense, a false positive coincides with "yes (positive) agriculture is

54 responsible for downstream pollution" when in reality this is not true (false).

55 Correspondingly, a false negative decision is committed when answering "no

56 (negative) agriculture is not responsible for downstream pollution" when in reality it

57 is responsible (false). In addition to the current situation, action programmes should

58 consider whether the nutrients issue is likely to increase or decrease in the future. In

59 this case the decision question "is upstream agricultural activity likely to become

60 responsible for downstream pollution in the next 7-10 years?" can lead to false

61 positives if action programmes address areas where the nutrients issue is neither

62 currently nor in the future likely to become important. In this context, false negatives

63 emerge when action programmes address areas where the nutrients issue is currently

64 very important and may likely remain so in the future (false negative). In any case, an

65 informative forecast of the future effects of agriculture on the environment can alert

66 policy to be ready to establish programmes or to respond by modifying the incentives

67 provided in existing programmes.

68   The aim of this paper is to propose an integrated decision making framework

69 for designing and establishing AEMs targeting nutrient reduction. This decision making

70 framework reduces the risk of committing false positives and wasting financial

71 resources or the risk of committing false negatives and not protecting the

72 environment. Section 2 of this work, briefly reviews the sources contributing to the

73 risk of committing either false positives or false negatives and sketches the proposed

74   decision making processes. Section 3 presents the Greek case study of the Louros

75   watershed and describes the methods, information sources and underlying

76   assumptions in the derivation of the various alternative scenarios associated with the

77   adoption of agri-environment programmes, CAP reform and climate changes affecting

78   both the hydrology of the catchment and the nutrient uptake rate of plants. Section 4

79   presents the results of the analysis, while section 5 concludes and draws policy

80   recommendations for a safer decision making process during the design and

81   implementation phases of AEMs.

82

83   **2. Sources of False Positives and Negatives in the Design of Agri-environmental**

84   **Policy**

85   Mandatory and voluntary AEM aim, amongst others, to reduce nutrient

86   concentrations in downstream rivers, lakes and wetlands. Most frequently, such

87   measures directly target nutrient deposition (inputs) to land by setting maximum

88   application rates. For example, the Nitrates Directive states that the amount of

89   livestock manure applied on agricultural land each year, including that applied by

90   animals themselves, should not exceed a maximum of 170 kg of nitrogen per hectare.

91   Other measures attempt to manage nutrients on the field, by promoting favourable

92   farm practices such as crop rotation systems, while others aim at restricting leaching

93   of nutrients from the field, through (e.g.) the maintenance of buffer strips. The design

94   and implementation of agri-environment action programmes for nutrient control is

95   based on information about nutrient deposition from agricultural and livestock activity

96   measured in kg per hectare and the concentration of nutrients in surface and

97   groundwater measured in mg/L. This practice of setting policy targets presumes a

98   direct relationship between nutrient deposition on the field and downstream nutrient

99   concentration. As such, it fails to take account of the static abiotic environment

100   (geology and soil) and the dynamics of human activity and climate change, as well as

101   of the changing and fluctuating water supply. Thus, we should not presume a linear

102   and static relationship between deposition and concentration on which to build solid

103   and robust AEM.

104   Figure 1 below attempts to sketch how false positives and false negatives may

105   be generated in agri-environmental policy-making. The upper part of the diagram

106   provides a coarse picture of the nutrient deposition-leaching-transportation-

107   concentration process and how this process is influenced by abiotic, biotic, human

108   activity and climate change factors. Under abiotic factors we refer to those physical

109   processes pertinent to the geology, topography, soil physical and chemical properties.

110   Under biotic factors, we refer to the whole range of sources that contribute to nutrient

111   deposition such as land uses other than agriculture and animal activity other than

112   livestock and/or grazing. Under human activity factors, we refer to agriculture and

113   other activities contributing nutrients and including municipal and industrial sources

114   coming from septic tanks or other devices of establishments that are not connected

115   to municipal wastewater networks, animal wastes, food processing, etc. In addition,

116   activities other than agriculture, may have an impact on the hydrology and especially

117   on the quantity and frequency of water provided to water courses. Beyond irrigation

118   and its corresponding drainage networks, examples include water extraction for

119   municipal and industrial uses and sometimes small or large scale energy production

120   from hydro electrical power plants. Finally, climate and especially temperature and

121   runoff are important factors determining nutrient cycling and transport (Howarth et

122   al., 2012).

123   The relationship between agricultural inputs and instream nutrient

124   concentrations is not a simple one, for example because of a large groundwater store

125   that may act as nutrient reservoir or because water flows may change, or plant uptake

126   may increase, or land use may change (Jackson et al., 2008; Howden et al., 2010). Thus,

127   the underlying relationship between nutrient deposition by agriculture and its impact

128   on nutrient concentration in downstream water bodies may be important (yes-

129   positive) or not (no-negative). However, without an integrated approach modelling

130   the relationship between nutrient input and instream concentrations there will be

131   uncertainty as to whether the policy can address the input-output relationship

132   accurately, and therefore avoid the risk of false positives and false negatives. Table 1

133   provides an indicative list of false positive or negative decisions along with connotative

134   reasons causing these deceptive decisions. A similar table may be generated if

135   dynamic changes caused by land use and climate change are taken into account. In

136   this context, dynamics may generate an agricultural pollution issue in areas that

137   currently have not such an issue and vice versa.

138   In this work we focus on two broad areas within the policy design process

139   which may contribute to the generation of false positives and false negatives:

140   • Appropriate baseline monitoring and modelling of the nutrient deposition-

141   concentration function and the resultant baseline abatement function

142   measured in terms of nutrient concentration in the water downstream and,

143 • Forecasting and incorporation of changes resultant from human activity and

144 climate change and the resultant dynamic abatement function again measured

145 in terms of nutrient concentration in the water downstream

146 Taking into account the long-term horizon for implementing an agri-environment

147 programme, policy design, and especially baseline modelling, should consider dynamic

148 changes that may considerably alter the initial conditions that lead to the adoption or

149 the rejection of an agri-environment programme in a specific area. For example,

150 within a seven year agri-environment planning horizon, several changes may occur in

151 land use, in agricultural production or/and even climatic conditions. Land use changes

152 may be instigated by agricultural policy changes such as the CAP, which may lead to

153 the abandonment of agricultural production or to the drastic change in the adoption

154 of cultivations with different nutrient applications (Barbayiannis et al., 2011). One

155 vivid example is the decoupling of Pillar 1 subsidies, which in some EU areas, has

156 induced the abandonment of several cultivations or the shift to other crops, including

157 nitrogen fixing legumes and the consequent reduction in nitrogen deposition. At the

158 same time other, economy-wide developments, may affect (increase) agricultural

159 input prices resulting to a rationalization and the consequent reduction of nutrient

160 deposition.

161 In conventional policy design, targets are set on deposition, assuming that a

162 proportional reduction will be achieved in the corresponding concentration of

163 nutrients. The Nitrates Directive and several other EU, national and regional policies

164 set such targets. This approach promotes "one-size fits all" policy and fails to take

165 account of the aforementioned specificities of the environment and of human activity

166 in the target-area(s), that call for a case-specific and "tailor made" approach to agri-

167 environmental target setting. In this work it is suggested that the baseline situation

168 should be modelled according to an integrated framework accounting for dynamic

169 changes. In this respect we minimize the risk of false positives and false negatives. To

170 this end we advocate a procedure that uses a dynamic, mass-balance water quality

171 model to help explain the input (deposition) – output relationship and integrates

172 science and socio-economic models to protect policy design from committing false

173 positives or false negatives (Skuras et al., 2014).

174 Figure 2 depicts this approach in a sequence of policy design steps supported

175 by science and social science methods and models. Once the non-compliance issue is

176 recognized and defined (step 1) with the support of existing data and socio-economic

177 public participation models, an integrated model of nutrient and sediment

178 transportation within the catchment is proposed to be constructed (step 2). This step

179 is supported by scientific models of nutrients and/or sediment transport that calibrate

180 a baseline situation based on flow and hydrochemistry conditions of the catchment

181 depicted by meteorological, soil-geological, flow, land use and water quality data. In

182 step three, decision makers will have the capacity to avoid false negatives and false

183 positives. False positives are usually generated by failing to take into account the

184 whole range of sources contributing nutrients to the watershed and overestimate the

185 contribution and impact of agriculture. In this context, adopting a policy to control

186 nitrogen deposition from agriculture will not have an effect. Potentially, false positives

187 may be generated by situations in which high nutrient deposition fails to show up in

188 water nutrient concentrations for various reason including geology, e.g., extensive

189 carstic phenomena that redirect nutrient rich water to neighbouring watersheds or to

190   underground water reservoirs, soil conditions that favour high denitrification,

191   deposition at river banks, etc.

192   In steps 4 and 5, mitigation measures are proposed and their effect is examined

193   according to the calibrated baseline model. This will allow the examination of the

194   simulated effectiveness of the mitigation measures and hence, the prevention of false

195   positives, by adopting measures that will not be effective or the prevention of false

196   negatives, by rejecting measures that will be effective (step 6). In step 7 the baseline

197   condition and the mitigation measures are re-estimated and simulated against

198   changing conditions including climate, land use and production. This will allow the

199   prevention of false positives in the sense that a deposition-concentration situation

200   that seems positive today may be most likely ameliorated in the near future due to

201   changing conditions, without the need of mitigation measures and thus, adopting a

202   programme would be less appropriate (step 8). The same step will allow the

203   prevention of false negatives in the sense that a seemingly unrelated deposition-

204   concentration situation today may be most likely aggravated in the near future due to

205   changing conditions and adopting mitigation measures under an agri-environment

206   programme would be appropriate.

207

208   **3. Case Study and Methods**

209   *3.1. The case study area of the Louros watershed*

210   The Louros water catchment (926 km^2) is situated in the central-southern part

211   of the Epirus (NUTS 2 region) Water District in Greece. The river rises in the mountains

212   adjacent to the "Dodoni Oracle", one of the most important and famous oracles of

213   ancient Greece. The main river flows for 72 km, and its waters are derived from many

214   spring and snow fed tributaries. The river forms a delta where it empties into the

215   Amvrakikos Gulf, a site listed under the Ramsar Convention and the Natura 2000

216   network[1]. The river's delta includes freshwater marsh with the largest reedbeds in

217   Greece, wet meadows and seasonally inundated land, lagoons, barrier spits, a major

218   saltmarsh, and some of the most extensive tracts of riparian forest remaining in

219   Greece. The Amvrakikos Gulf is very important for biodiversity and a unique

220   biogeographical refuge in the migratory route between Europe and Africa. Located

221   deep into the Mediterranean, well-connected to the Balkans and the European

222   mainland serves as a bridgehead for mutliple migration routes from and towards

223   Africa with 182 bird species observed to breed, winter, or stage in the area. Of these

224   birds, 70 are listed in Annex I of Directive 79/409/EEC, detailing the species in need of

225   special conservation measures[2].

226   The river's annual discharge at its mouth is 95.13 m^3/s and the density of its

227   hydrographic network in the catchment is 0.69 km/km^2. Despite the operation of a

228   relatively small (10.3 MW) hydroelectric power plant, the river has continuous water

229   flow due to the serious siltation of the dam. The upper part of the catchment is

230   mountainous and semi-mountainous with the highest elevation at 1,976 m. The lower

231   part of the catchment is plain and Louros river, together with the adjacent Arachthos

232   river, irrigates and drains the most significant plain, in terms of agricultural production,

233   of Western Greece. The catchment receives relatively large volumes of convective

234   precipitation, and rainfall is high for Mediterranean conditions. The average annual

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[1] The site's description and map under the Ramsar Convention can be found at:
https://rsis.ramsar.org/ris/61 and under the Natura 2000 network at:
http://natura2000.eea.europa.eu/
[2] Last amendment of Annex I of Directive 79/409/EEC is found in Directive 2009/147/EC, Official Journal, L20/7 of 26.1.2010.

235   precipitation ranges from 800 mm per year in the lowlands up to 1300 mm per year in

236   the mountainous areas. Today, Louros provides drinking and industrial water to the

237   three largest urban areas of the catchment and many smaller towns. Farming, tourism,

238   stock raising, aquaculture at the uplands and fish farming at the estuaries are the most

239   important economic activities directly or indirectly dependent on the quality and

240   quantity of Louros' water.

241   Chemical analyses undertaken in several monitoring points indicate high

242   conductivity and concentrations of pollutants mostly in the estuary where drainage

243   channels return drained irrigation water. The river has been highlighted as vulnerable

244   for eutrophication, and two published studies have classified the water quality as

245   "fair" or "poor to fair" (Ovezikoglou et al., 2003; Kotti et al., 2005). However, nutrient

246   concentrations are recorded in relatively low levels in the published studies (average

247   nitrate < 1 mg-N/L, average phosphate < 15 µg-P/L). Maize, medic (clove) and cotton

248   are the most widely spread irrigated arable cultivations with considerable fertilization,

249   while wheat is mostly rain-fed with minimum fertilization. Citrus fruits, mainly orange

250   and mandarins and to a less extent lemon trees and kiwi fruits are the most important

251   irrigated perennial cultivation while olive groves are mostly rain fed with minimum or

252   no use of fertilizer.

253   Extensive consultation with the scientific community and stakeholders

254   concluded that the almost 10,000 ha of intensively cultivated land within the

255   watershed (3,300 ha of maize; 4,000 ha of medic; 340 ha of cotton; and 2,100 ha of

256   citrus fruit) contribute an average annual of about 1,780 tons of nitrogen and 1,160

257   tons of phosphorous in terms of deposited active substance. Local scientists and

258   stakeholders were presented with average national estimates of fertilization per

259 cultivation and per hectare. Then, they were asked to adapt them (increase or

260 decrease) for the local corresponding cultivations and provide the reasons why they

261 suggested adaptations. For example, maize cultivations, depending on the site and soil

262 properties, accept during starter band fertilization an average of 1,000 kg of fertiliser

263 with a NPK ratio of 16-20-0 were applied per hectare corresponding to 160 Kg of N, or

264 1,000 Kg of 18-12-8 fertiliser corresponding to 180 Kg of N per hectare. During surface

265 fertilization the same plots accept usually 300 to 350 Kg of 25-0-0 fertiliser per hectare

266 corresponding to 75 to 90 Kg of N. The corresponding phosphorous fertilization is

267 about 120-200 Kg per hectare depending on soil needs by using either 18-12-18 or 16-

268 20-0 fertiliser during starter band fertilization. Application of phosphorous during

269 surface fertilization is rare in the case of maize.

270 Water chemical analyses carried out by the Greek Ministry of Rural

271 Development and Food (MRDF) showed that at locations close to the estuary, the

272 concentration of nutrients (nitrates, ammonium and total phosphorous) was relatively

273 high at least during autumn and early winter. Following this rather weak evidence, in

274 2006 MRDF established the plains of the Louros catchment and part of the adjacent

275 Arachthos catchment, as a Nitrification Vulnerable Zone (NVZ) under the Nitrates

276 Directive. The plan that, as yet, has not been implemented due to Greece's financial

277 crisis and consequent budgetary constraints, allows compensation for farmers of the

278 aforementioned cultivations if they comply with measures or combination of

279 measures including the set aside of land, maximum allowable fertilization levels and

280 irrigation per cultivation.

281

282

3.2. Baseline Modelling

283

284   In this work, the INCA-N and INCA-P integrated catchment models were used

285   to simulate the distribution of nitrogen and phosphorous correspondingly in the

286   aquatic and terrestrial environment. The models can simulate the annual and seasonal

287   variations in the stream-water concentrations of nitrate (NO3), ammonium (NH4),

288   total phosphorous (TP) and dissolved phosphorous (DP) (Wade et al, 2002a, 2002b).

289   The models take account of anthropogenic nutrient inputs in the form of fertiliser or

290   sewage discharges on top of natural nutrient inputs through atmospheric deposition,

291   vegetation and mineralisation (and subsequent nitrification). Figure 3 shows the

292   flowchart of calibrating the INCA-N baseline model for the Louros watershed and of

293   using the model for simulating the effects of mitigation measures and the effects of

294   future climate and land use changes. A similar flowchart holds when applying the

295   INCA-P model. The quality and quantity of data Inputs is the most crucial stage in

296   calibrating the baseline models (Figure 3). The Louros catchment was divided into 16

297   smaller reaches (or sub-catchments), according to where observations of chemistry or

298   flow are available, a procedure that is considered standard for semi-distributed

299   models (Whitehead et al., 1998). The groundwater recharge area of the Louros is

300   considerably larger than the topographic catchment due to the extensive karstic

301   formations. For each of the 16 sub-catchments, daily temperature and precipitation

302   data were estimated from the three meteorological stations situated in or around the

303   catchment, weighted using Theissen polygons. Detailed land cover for six major

304   classes was provided by CORINE.

305   For each crop, scientific sources and communication with expert agronomist

306   in the area were utilised to calculate average deposition rates for each nutrient and

307 their approximate time of application. In addition, information was collected for the

308 irrigation water needs of each crop. These calculations were presented to

309 stakeholders during locally organized workshops and were fine tuned for various

310 cultivation practices, micro-environments and multinutrient fertilisers. Nutrients

311 applied through manure were estimated from annual statistical records assuming an

312 average output per type of grazing animal. Hog farming depositions were also

313 calculated as point source pollution directed to river reach.

314 Biological fixation of nitrogen was included as an extra source of nitrogen for

315 non-arable land use classes. This was assumed to equal 4 kg-N/(ha·year) for shrub

316 land, and 10 kg-N/(ha·year) for forests. For phosphorus, the respective quantities

317 were at a rate of 1 kg-N/(ha·year) for shrubland and of 2 kg-P/(ha·year) for forests.

318 Finally, annual atmospheric values of dry and wet deposition of nitrate and

319 ammonium were calculated per European Monitoring and Evalution Programme

320 (EMEP) grid square (Cooperative Programme for Monitoring and Evaluation of the

321 Long-range Transmission of Air Pollutants in Europe). The Louros catchment is covered

322 by three different EMEP squares. Wet deposition was separately calculated for

323 forested and non-forested land cover. A 50-50 split between nitrate and ammonium

324 was assumed for nitrogen addition while for phosphorus, 70 % was assumed to be

325 added as solid P, and 30 % as liquid P.

326 To model the hydrology, the hydrological model PERSiST (Precipitation,

327 Evapotranspiration and Runoff Simulator for Solute Transport) was used (Futter et al.,

328 2013) to generate hydrological input data (Hydrological effective rainfall and Soil

329 moisture deficit) needed to drive the chemical INCA models. The model was set up

330 with the aforementioned six land cover and three different soil boxes, i.e., one quick

331 box, one soil box and one groundwater box. Each box is characterized by nine different

332 parameters, which are specific for each land class. There are nine additional land cover

333 specific parameters, related to properties such as snow melt, evapotranspiration and

334 base flow index. PERSiST was calibrated against the observed flow for the period Jan

335 2001 – Sep 2012. The INCA-N parameters calibrated included soil denitrification, soil

336 nitrification, soil mineralisation, plant NO_3 uptake, plant NH_4 uptake, in-stream

337 nitrification, in-stream denitrification, initial groundwater nitrate and initial

338 groundwater ammonium. The size of the point source (effluent concentration of

339 ammonium) was also calibrated, the hydrological parameters of groundwater

340 residence time was adjusted to improve the fit for base-flow conditions, and the

341 drought runoff fraction was adjusted to keep more nitrogen in the soil during the dry

342 summer months. The INCA-P parameters calibrated included soil phosphorus terms

343 (Freundlich isotherm, weathering factor, sorption coefficient and equilibrium

344 phosphorus concentrations), plant uptake, process rates response to temperature,

345 immobilisation, initial labile and inactive soil P, reach ecology parameters for

346 macrophytes and epiphytes, and groundwater phosphorus terms. Both models, INCA-

347 N and INCA-P were calibrated against nutrient concentration data from monitoring

348 stations operated by the Ministry of Environment and Energy with reasonably good

349 overall results and goodness of fit measures.

350

351 *3.3. Simulating mitigation measures and future changes*

352 In this section we detail the processes for simulating the effects of mitigation

353 measures and the effects of future climate and land use changes on the baseline. In

354 Figure 3, once the baseline simulation has been calibrated, we decide whether

355   mitigation measures are needed. If mitigation measures are introduced, their effect

356   on reducing nutrients loads is simulated on the lower left part of the Figure 3. This is

357   an iterative process up until compliance is achieved, because the proposed mitigation

358   measures may not be effective. Once we end up with a set of effective mitigation

359   measures, or if no mitigation measures are needed, we examine whether the

360   preferred mitigation measures remain effective under future climate and land use

361   scenarios. This simulation exercise is shown on the lower tight part of Figure 3. At the

362   end of this process the decision maker will have enough and strong evidence for the

363   effectiveness of the proposed mitigation measures and their resilience to future

364   climate and land use changes and adequate information to establish a coherent

365   monitoring system. If more than one alternative mitigation measures comply with the

366   thresholds and are resilient to future climate and land use changes, then the decision

367   maker will be able to choose the most cost-effective.

368   In order to simulate the adoption of an agro-environmental programme we

369   considered mitigation measures that have been introduced in other NVZs during the

370   implementation of the 2007-2013 Rural Development Programme in Greece. For

371   annual cultivations the proposed agro-environmental scheme (Mitigation 1) includes

372   a composite scheme with 5% of the total land occupied by non-cultivated margins,

373   20% of the land under rotation with nitrogen fixing legumes, 20% of the land under

374   half of the standard fertilization scheme, and 25% reduction in the deposition of

375   fertilizers to the rest 55% of the land. This scheme achieves 51.25% reduction in

376   fertilization deposition in relation to the baseline. For each one of the major annual

377   cultivations (maize, cotton and medic), the actual deposited quantities of fertilizers

378   are calculated and subtracted from the total deposition in each sub-catchment.

379   Furthermore, adaptations are made to take account of uncultivated margins by

380   reducing the leaching coefficient. The proportional reduction in irrigation water is also

381   simulated. For perennial plantations of citrus fruit only a 25% reduction in deposited

382   fertilizers is considered. The same measure was simulated with increased reductions

383   to 30% (Mitigation 2). In this scheme, the cultivated land is distributed to 5%

384   uncultivated margins, 25% under nitrogen fixing legumes, 25% with half the

385   fertilization and the rest 45% of the land with a 30% reduction in fertilization and

386   irrigation. For citrus fruit plantations a 30% reduction in fertilization is envisaged.

387   For each cultivation, the mitigation measure cost was calculated based on

388   Standard Gross Margins (SGMs) provided for the region of Epirus by Eurostat's FADN,

389   the Farm Accountancy Data Network. The standard Gross Margin (SGM) of

390   acultivation is defined as the value of output from one hectare less the cost of variable

391   inputs required to produce that output. We assume that agro-environmental policies

392   induce only temporary changes to farm practices and thus, the constant cost of fixed

393   assets such as capital, land, and buildings is not affected and should not enter the cost

394   calculations.

395   Meteorological data from three different climate models were used to define

396   the meteorological time series for the 2031-2060 scenario-period, namely the KNMI-

397   RACMO2-ECHAM5 (abbreviated thereafter as KNMI), the SMHIRCA-BCM (abbreviated

398   thereafter as SMHI) and the HadRM3-HadCM3Q model (abbreviated thereafter as

399   Hadley) (Christensen et al., 2009). Observed meteorological time series were adjusted

400   by the average difference between the control and scenario periods for each month

401   and for each of the three climate models, as more sophisticated methods

402   (downscaling with a power function) resulted to very unrealistic precipitation

403 amounts, especially for summer months. The predicted relative changes in

404 precipitation did not differ substantially among the three climate models, with the

405 least change predicted by the KNMI model (-12 %) followed by the SMHI model (-14%)

406 and the Hadley model (-16 %). The seasonal patterns in precipitation change were

407 seemingly random, except for the month of July for which all three climate models

408 predict a large decrease in precipitation (55-65%). As concerns temperature, the three

409 climate models were more different, with the SMHI model predicting the smallest

410 increase (+1°C on average), the Hadley model predicting the largest increase (+2.2°C),

411 and the KMNI predicting an intermediate decrease(+1.8°C). Seasonal patterns are also

412 more pronounced, with a smaller increase in winter temperatures and a larger

413 increase in summer temperatures. The modelled climate change effects have an

414 impact on the hydrology of the area. For example, in one of the central and most

415 important reaches of the river, the simulated flow for the control period 1981-2010

416 was 16.6 m^3/s and decreased by 14.9 %, 18.3% and 27.7% under the KNMI the SMHI

417 and the Hadley models respectively. There is an even greater effect on the annual

418 minimum flow. For the same reach, the average annual minimum flow for the control

419 period was 8.1 m^3/s and is decreased by 20.6 %, 5.5% and 29.3% for the KNMI the

420 SMHI and the Hadley models respectively.

421 Climate change will also induce long term land-use and plant productivity

422 changes depending on the IPCC storyline (Nakicenovic et al., 2010). In general, the

423 IPCC storylines refer to the 'A'-scenarios representing a market-oriented future and

424 the 'B'-scenarios representing a more environmental-oriented future. Furthermore,

425 the '1'-scenarios represent a future globalised world, whereas the '2'-scenarios

426 represent a world with stronger national or local regulations. These storylines are

427   combined to produce various scenarios, e.g., A1, B1, A2, B2, and their variants. These

428   scenarios have direct impact on forecasted CO_2 concentrations. In 2009, the Bank of

429   Greece set up the "Climate Change Impacts Study Committee" with the mandate to

430   draft a report presenting the foreseen environmental, economic and social impacts of

431   climate change and estimating the cost of these changes for the Greek economy as

432   well as the cost of the proposed adjustment measures (Zerefos et al., 2011). In this

433   study, climate change impacts on agriculture have been measured for each of 11

434   Greek climate zones. The researchers used the AquaCrop (version 3.1, 2010) model

435   developed by the FAO (Doorenbos and Kassam, 1979) for modelling crop production

436   in relation to water (especially for rain fed cultivations) and an estimated higher

437   production response under increased concentrations of CO_2 and less production

438   under the risk of severe climate phenomena and diseases.

439   These forecasts have been combined with simulated hydrology changes in the

440   area to produce alternative broad land use and fertilization changes. For the Louros

441   watershed, the highest negative change is projected for wheat (almost -10% of land

442   area planted) and the highest positive change is projected for cotton, vineyards and

443   olive groves (almost +10%). The fertilization for all other cultivations either remains

444   unchanged or is projected with minor (less than 5%) negative or positive changes. The

445   aforementioned IPCC storylines are combined with climate change model forecasts to

446   produce alternative combinations of long term climate and land use changes. In the

447   case of the Louros watershed, the scenario with the least impacts, called thereafter

448   the "best" future scenario, is the KNMI model combined with the B1 storyline and the

449   scenario with the most severe impacts, called thereafter the "worse" future scenario

450   comes from the Hadley model combined with the A2 storyline.

451   In the short-term land use changes may be induced by the continuous CAP

452   reforms. The CAP, since 2005, has gradually moved away from coupled payments

453   towards more decoupled payments. Greece adopted the so called historical model

454   that calculated decoupled payments based on historical records of production and

455   subsidies. For many products, decoupled subsidies are granted almost unconditionally

456   with the only obligation being for the farmers to take care of the good ecological status

457   of the land. For cotton, 65% of the subsidy is decoupled and 35% depends on

458   delivering a minimum amount of cotton. This has affected both the area used for

459   cotton and the amount of deposited nutrients. The land under cotton has decreased

460   dramatically especially by farmers who choose to take the decoupled part and switch

461   cultivation or leave the land uncultivated. The farmers targeting both the decoupled

462   and the coupled parts of the cotton subsidy do not aim to maximize production but to

463   minimize costs, including cost for fertilization, because the minimum production

464   allowing the farmer to qualify for the coupled part of the subsidy is very low and can

465   be attained with minimum inputs. In the period following decoupling (2005-2009) the

466   area cultivated by cotton was reduced by almost 40% and the area under wheat by

467   almost 30%. Taking into account the Commission's decision to continue this trend for

468   further decoupling and the new binding "Greening" rules for 2014-2020, we assumed

469   that an amount of marginally fertile land cultivated by cotton and maize will be

470   withdrawn and a reduction of fertilization will take place within a wider farm survival

471   strategy to reduce operating costs.

472   After extensive consultation with the scientific community and local

473   stakeholders, it was decided to model land use change due to the changing agricultural

474   policy and markets for agricultural products as a 25% set aside for capturing those

475   farmers who will stop farming with the decoupled part of the subsidy and a 25%

476   reduction in the use of fertilizers for those farmers who will continue cultivation,

477   aiming to the coupled part of the subsidy. This land use scenario was translated to

478   reduced nutrient deposition per cultivation and sub-catchment because certain

479   cultivations such as cotton are highly localized within the watershed. Following the

480   suggestions from the projections of local stakeholders, we escalated the same land

481   use projection to 30% reduction in land cultivated and fertilizer used and run both

482   simulations.

483

484   **4. Results**

485   *4.1 The occurrence of a False Positive*

486   Taking into account only the supply of nutrients, and especially those from

487   agricultural activity, it is estimated that the watershed accepts an amount of 2,594

488   tonnes of active N substances and 1,578 tonnes of active TP per annum from which

489   agriculture is responsible for almost 1,780 tonnes of N and 1,163 tonnes of P for the

490   major cultivations within the watershed. These amounts of active fertilizer substance

491   alone are enough to trigger public concerns over agricultural activity in relation to the

492   high nature value of the lagoon and its importance for European biodiversity, despite

493   the fact that monitoring data were sparse and showed at most moderate nutrient

494   concentrations and few signs of eutrophication. The simulated average and monthly

495   concentrations for nitrates, ammonium and Total Phosphorous (TP) are shown in

496   Figure 4.

497   Simulated nitrate concentrations near the estuary range between 0.8-1.0 mg

498   N/L with an average at 0.9 mg N/L, while ammonium concentrations range from 0.04-

499 0.13 mg N/L with an average of 0.08 mg N/L. TP concentration ranges from 0.02-0.11

500 mg TP/L with an average of 0.05 mg TP/L and SRP concentration from 0.01-0.11. mg

501 SRP/L with an average of 0.04 mg SRP/L. Skoulikidis et al (2006) have proposed a

502 Nutrient Classification System (NCS) for small/medium sized rivers in Greece based on

503 annual average concentrations from 36 sites throughout Greece. According to this

504 system, the river is classified as of moderate quality in relation to nitrates (0.6-1.3 mg

505 N/L) and ammonium (0.06-0.20 mg N/L) and of high quality in relation to TP (0.17-0.22

506 mg TP/L). Under other classifications, e.g., the nutrient quality classes in French and

507 Italian rivers (Skoulikidis et al., 2006), the Louros river would be placed between a

508 "Good" and "Moderate" class. At the same river and sub-catchments, Macrophyte

509 data (taxon name and abundance class) were collected and the IBR (Indice Biologique

510 Macrophytique en Rivière - Macrophyte Biological Index for Rivers) was calculated by

511 Manolaki et al (2011) according to the methodology proposed by Haury et al. (2006).

512 Of the 17 sites they studied, eight are characterized as having "High" ecological status,

513 three as "Good", four as "Moderate" and two as "Poor". The best predictors for the

514 decrease in IBMR values were salinity and water temperature, while SRP was also

515 found to be correlated with IBMR but able to explain only 47 % of the variability in

516 IBMR values. The classification of the river's estuaries based on the aforementioned

517 simulated results was re-confirmed in 2013 by the Management Plan drawn for Epirus'

518 water resources.

519   Thus, assuming that there is a direct positive relationship between agricultural

520 activity and nutrient concentration would be a false positive, i.e., assuming a direct

521 relation that does not exist. This further supported by the fact that nitrate concentrations

522 tend to be highest in the upper reaches, which are not affected by agriculture, while

523 the poor ecological status for macrophytes can obviously not be attributed to nutrient

524 concentrations. There are several alternative explanations of why nutrient deposition

525 rates do not really contribute to high nutrient concentrations downstream. As local

526 stakeholders argue, due to cost minimization strategies and the rising price of

527 fertilizers and energy, farmers take very good care of the time of fertilizer application,

528 of the appropriate amount of fertilizer and of irrigation. This may contribute to a more

529 balanced nutrient deposition and nutrient uptake by plants leaving less residual

530 nutrients on the soil. In the framework of cost minimization there is also reduced and

531 more precisely applied irrigation for reducing the cost of energy. Thus, higher uptake

532 by plants also may be supported by longer water residence time in the soil brought

533 about by more modern irrigation schemes (drop irrigation) that are gradually replacing

534 sprinklers. This practice also reduces leaching and nutrient transportation.

535   Finally, there are well documented physical and biological processes that may

536 contribute to lower nitrogen levels despite higher deposition rates. Denitrification and

537 nitrogen immobilization in excess of mineralization, at least temporarily when

538 temperature is high and the concentration of soil C is high (Saggar et al., 2013). High

539 spring and summer temperatures enhance aerobic respiration and denitrification

540 while aerobic respiration further enhances denitrification by consuming oxygen,

541 resulting in strong sensitivity of denitrification to temperature though substrate type

542 and soil moisture may limit microbial processing (Butterbach-Bahl et al., 2013; Luo et

543 al., 2013). Finally, sediment and thus nutrient transportation has been reduced in the

544 area due to the extensive drainage and river bank stabilization works that have been

545 undertaken throughout the watershed in the last 30 years.

546

547 *4.2 Agri-environmental Measures and Future Changes*

548   The simulated effects of agri-environmental measures, climate change, land

549 use change and their combinations on nutrient concentration at the reach nearest to

550 the estuaries of Louros river are presented for the simulated concentrations of nitrate,

551 ammonium, SRP and TP in Table 2. If the mitigation measures described in section 3

552 of this work are adopted, the simulated reduction in average nitrate and ammonium

553 concentrations is not significant. Quantitatively, exactly the same changes can be

554 effected by short-term land use changes induced by the CAP without any mitigation

555 measure. Furthermore, as concerns long-term climate change impacts, even the effect

556 of the worse scenario does not show any important impact on nitrates and ammonium

557 concentrations. The imperceptible modelled net change in nitrate concentrations for

558 the climate scenarios is due to the fact that the amount of nitrogen leaching from the

559 soils decrease by approximately the same rates as the runoff. At the same time, the

560 average discharge decreases by between 15 and 28 %, the amount of nitrogen

561 leaching from the soils decreases by 15 % (KNMI and SMHI models) to 25 % (Hadley).

562   The amount of nitrogen transported to the estuaries is however substantially

563 reduced, by 16.3 % for the KNMI climate, 17.0 % for the SMHI climate, and by 26.5 %

564 for the Hadley climate. The main reason for the simulated decline in nitrate leaching

565 is that longer water residence time in the soil and stream and less runoff meant that

566 more of the nutrients were available for plant uptake which balances the additional

567 fertiliser load under increased CO_2 concentrations. Furthermore, due to lower

568 atmospheric deposition, the external loads were around 5 % lower for the climate

569 change scenarios.

570   The simulated effects resultant from the different scenarios are more

571   significant on phosphorous than on nitrates and ammonium. Mitigation measures

572   reduce SRP from 27.3% (Mitigation1) to 30.1% (Mitigation 2) and TP from 24.3%

573   (Mitigation 1) to 27.2% (Mitigation 2). This reduction can, for sure, classify the river to

574   the "Good" status as concerns phosphorous. The same results are achieved by the

575   scenario of short-term land use change induced by CAP. As concerns the sole effect of

576   climate change, no significant changes are observed. The simulated response of

577   phosphorus concentrations to climate change is mainly due to a combination of

578   decreased leaching due to higher removal rates from the soil brought by longer soil

579   water residence times, and less dilution due to reductions in flow. The amount of SRP

580   leaching from the soil decreased by 16.9 % (SMHI), 18.7 % (KNMI) and 35.6 (Hadley).

581   The amount of SRP transported to the estuary is reduced by 31.1 % (SMHI), 34.4 %

582   (KNMI) and 46.6 % (Hadley). Although results from the three climate models differ

583   somewhat, the tendency is that the increase of phosphorus concentrations will be

584   more pronounced during summer months, whereas they will remain unchanged or

585   even decrease during the winter months. The month of July stands as an exception to

586   this pattern, for which phosphorus concentrations remained nearly unchanged. This

587   may be attributed to the forecasted low precipitation, which results in less phosphorus

588   leaching from the soil.

589   From the aforementioned discussion it is clear that, if mitigation measures are

590   adopted in order to upgrade the status of the river's water quality to "Good", at least

591   as concerns nitrates and ammonium, their effect is weaker than the effect that can be

592   achieved by the short-term land use changes observed and envisaged under the

593   reformed CAP. Thus, accepting that mitigation measures will be able to upgrade the

594   river's water quality as concerns nitrates to "Good" and comply with WFD's

595   requirements, will be yet another false positive decision in the design of agri-

596   environmental policy. A false negative may emerge if decision makers fail to recognize

597   the effect of short-term land use changes at least on phosphorous. Climate change

598   does not seem to exert an important positive (best scenario) or negative (worse

599   scenario) effect. In all scenarios, TP is reduced, even slightly. Recent evidence shows

600   that phosphorus can determine river phytoplankton growth irrespective of the

601   nitrogen concentration (Wang and Wang, 2009) and the physical conditions of light,

602   water temperature and residence time are important in lowland river catchments.

603

604   *4.3. The Cost of a False Positive*

605   For each one of the four major cultivations in the watershed the cost of the

606   mitigation measures was estimated. In order to proceed in our calculations we carried

607   out two focus groups with stakeholders and elite interviews with agronomists in the

608   area. Farmers' income from the different cultivations was estimated from the

609   Standard Gross Margins derived by the FADN database for the region of Epirus where

610   the Louros watershed is situated. From the FADN database we also calculated initial

611   estimates of the cost of fertilization, and the cost of cultivating lentils, as well as the

612   SGM of the lentil for fodder. Elite interviews with agronomists were utilised to

613   estimate the loss in production due to reduced fertilization and irrigation.

614   Consequently, stakeholders were presented with the initial estimates during a focus

615   group with the aim to discuss and adapt initial estimates of the exact effects of

616   reduced fertilization and irrigation on production and on farmer's income. In the

617   context of this focus group, the transaction cost for submitting an environmental plan

618   and subscribe to an agri-environmental programme were also estimated. The cost of

619   the mitigation measures for each one of the four major cultivations was estimated as

620   income forgone from reduced production plus transaction costs minus cost avoided

621   from reduced fertilization and irrigation and fodder production. For citrus fruit

622   plantations only income forgone was estimated, as there is no way to have land under

623   set-aside.

624   The average cost estimates for abating nitrates and TP for the different

625   cultivations in the area and the watershed as a whole, should the Mitigation 1 scheme

626   be adopted by all farmers located within the hydrological boundaries of the

627   watershed, is presented in Table 3. The upper part of Table 3 provides average cost

628   estimates for fertilizer reduction per hectare (ha) and kilogram (kg) of active substance

629   for the four major cultivations and the watershed as a whole. The cost per hectare

630   varies significantly from 437.2 €/ha for the less profitable cultivation of medic to 657.2

631   €/ha for the most profitable cultivation of cotton. The cost of abating one Kg of pure

632   nitrogen ranges from 4.5 €/kg for corn to 12.5 €/kg for medic. For phosphorous, the

633   cost of abatement per Kg is much higher than for nitrogen ranging from a high of

634   almost 54 €kg for citrus fruit cultivation to a low of 5.4 €/kg for corn.

635   These estimates can be compared with past estimates of abatement costs for

636   seven EU Member States carried out in the framework of a study estimating the ex

637   post costs of implementing the Nitrates Directive in Europe (Kuik, 2006). In this study,

638   cost estimates at 2004 prices range from a high 236 €/ha in the Netherlands to a low

639   of 6 €/ha in the UK, which, however, refer to livestock and grasslands respectively. As

640   concerns the cost per Kg this range from a low 0.4 €/kg for Croatia, then not a member

641   State of the EU, to a high of 3.5€/kg for the Netherlands. Taking into account that

642   these estimates were derived with the Nitrates Directive in focus, they refer to

643   grasslands and livestock which are not as intense activities as, for example, cotton.

644   They also target a nitrogen concentration of 50 mg/l set up by the Nitrates Directive

645   for sub-surface waters. In our study, the nutrient loads are already low and thus the

646   marginal abatement cost is at its steeply rising part. An indirect way to measure

647   abatement cost is through prohibitive standards, penalties and/or taxes. In the

648   Netherlands, between 1998 and 2005, penalty-free thresholds were gradually

649   reduced – for example, for nitrogen from 300 kg/ha to 140 kg/ha for grassland farms

650   (Goffe, 2013). Penalties, in the Netherlands were fixed at €0.68/kg for nitrogen and

651   €2.60-€10.40/kg for phosphorous in 1998, and were increased to €2.53-€5.07/kg and

652   €20.60/kg respectively (Goffe, 2013) while levy taxes in 2003 were set to 2.3 €/Kg for

653   nitrogen and 9.1 €/Kg for phosphate (Söderholm and Christiernsson, 2008) which

654   compare with the results of our study.

655   The focus of this study, however, is to reveal the high abatement cost when

656   this is measured in terms of reduced nutrient concentration downstream. The cost

657   estimate for nitrates is unreal at the unprecedented levels of just over 300 thousand

658   euro per reduced microgram per litre €/[(µg/l)]. For phosphorous this is at 412,398

659   €/[(µg/l)]. So, a false positive decision to comply with WFD and attain a "Good" status

660   as concerns nutrient loads would be obviously unacceptable by any taxpayer in

661   Europe. Which are the reasons for this case? First, the nutrient status is already at

662   "Moderate" to "Good", i.e., the nutrient concentration is already low compared to the

663   50 mg/l threshold of the Nitrates Directive. Thus, the marginal cost to attain an even

664   lower level of concentration is extremely high. Second, at this level of concentration,

665   the simulations showed that even the withdrawal of 30% of the cultivated land will

666   not reduce nitrate concentration by more than 0.02 mg/l and TP concentration by 0.01

667   mg/l. These are negligible achievements at a highly disproportional cost.

668   To summarize the discussion so far, it can be argued that the abatement cost

669   of agri-environmental programmes aiming to manage nutrient loads should be

670   measured as a change in nutrient concentration and not at levels of abated fertilizer.

671   In other words, the targets of such agri-environmental programmes and policies

672   should be set at nutrient concentration levels and not at quantities of abated

673   substance either in mineral fertilizer or in manure and slurry. This can be attained if,

674   during the design of agri-environmental programmes, the status quo (baseline), the

675   impacts of the mitigation measures and the impacts from likely future changes are

676   simulated. Then, false positives and false negatives can be avoided, the cost-

677   effectiveness of mitigation measures can be assessed and an appropriate monitoring

678   system can be set up.

679

680   **5. Conclusions**

681   The EU's agri-environmental policy is a response to the growing public concern

682   over the environmental impacts of agriculture. As such, agri-environmental policy

683   attempts to meet requirements from the WFD, the Nitrates Directive and the Habitats

684   Directive, the cornerstone of environmental conservation in Europe. Agri-

685   environmental policy has grown to a tremendous budget (€36.6 billion spent in the

686   2007-2013 programming period across the EU) and power by affecting almost a

687   quarter of the EU's utilized agricultural area. This work concerned only with

688   programmes managing nutrient loads in freshwaters and not with other forms of agri-

689   environmental programmes. Results showed that, under public pressure and

690   seemingly high rates of mineral fertilization, decision makers may falsely adopt an

691   agri-environmental programme that may be both, ineffective in reducing nutrient

692   loads and cost inefficient. Furthermore, they fail to take account of future changes

693   that may inactivate the proposed mitigation measures, aggravate or reverse the

694   baseline situation.

695   The present work suggests that the design of such agri-environment

696   programmes should evolve to a thoroughly designed, interdisciplinary exercise

697   integrating science and social-science models in a step-wise procedure. This process

698   will ensure decision makers with the highest possible information from scientific

699   sources and models and from local knowledge. This information can be used by

700   appropriate simulation models to calibrate the baseline situation. Once an

701   appropriately calibrated model is derived, further scenarios simulating policy, land use

702   and climate changes can be simulated. Based on these results the effectiveness and

703   cost efficiency of the proposed actions and of envisaged changes can be assessed. As

704   a result, decision makers will be able to grasp an ex-ante evaluation of the current

705   situation and of the proposed mitigation actions, if needed. This will allow decision

706   makers to monitor the current situation and respond by adopting new measures or

707   adapting existing ones to the changing physical, social and policy environment.

708   Under this proposal, the cost of the design phase of an agri-environmental

709   programme will increase. But, in view of the high cost of mitigation measures, such an

710   increase in the design stage of the agri-environmental policy should be considered as

711   an insurance against the commitment of very expensive false positive and false

712   negative decisions. Finally, in this work it is proposed that the targets of agri-

713   environmental policy and consequently, the measurement of abatement cost should

714   be done in terms of nutrient concentrations and loads in water and not in terms of

715   physical quantities of abated substance in the field. This will provide the cost efficiency

716   exercise with a wider perspective as concerns the sources of nutrients and abiotic,

717   biotic and anthropogenic activities that contribute the nutrient loads. In turn, this will

718   force agricultural policy decision makers to coordinate their actions with other

719   environmental policy makers for achieving maximum results and avoiding internal

720   contradictions.

721

729 **References**

730

731 Barbayiannis, N., Panayotopoulos, K., Psaltopoulos, D. and Skuras, D. 2011. The

732   influence of policy on soil conservation: A case study from Greece. Land

733   Degradation and Development 22(1), 47-57. DOI: 10.1002/ldr.1053

734 Batáry, P., V. Dicks, L., Kleijn, D., Sutherland, W. 2015. The role of agri-environment

735   schemes in conservation and environmental management. Conservation

736   Biology, 29(4), 1006–1016. DOI: 10.1111/cobi.12536

737 Buckley, C., Wall, D.P., Moran, B., O'Neill, S., Murphy, P.N.C. 2016. Farm gate level

738   nitrogen balance and use efficiency changes post implementation of the EU

739   Nitrates Directive. Nutrient Cycling in Agroecosystems, 104(1), 1–13. DOI:

740   10.1007/s10705-015-9753-y

741 Butterbach-Bahl, K., Baggs, E. M., Dannenmann, M., Kiese, R. and Zechmeister-

742   Boltenstern, S. 2013. Nitrous oxide emissions from soils: how well do we

743   understand the processes and their controls? Philosophical Transactions of the

744   Royal Society B-Biological Sciences, 368, 1-13. DOI: 10.1098/rstb.2013.0122

745 Christensen, J.H., Rummukainen, M., and Lenderink, G. 2009. Formulation of very-

746   high-resolution climate model ensembles for Europe. Chapter 5 (pp.47-58), In

747   Van der Linden, P. and J.F.B. Mitchell (eds), ENSEMBLES: Climate change and

748   its impacts: Summary of research and results from ENSEMBLES project. Met

749   Office Hadley Centre, FitzRoy Road, Exeter EX1 3PB, UK. 160 pp.

750 Doorenboos, J. and A.H. Kassam (1979), "Yield response to Water", Irrigation and

751   Drainage Paper No. 33, FAO Rome Italy, 193 pp.

752   European Economic Community. 1991. Council Directive of 12 December 1991

753   concerning the protection of waters against pollution caused by nitrates from

754   agricultural sources. Official Journal of the European Communities, L 375, 1-8.

755   Fowler, D., Steadman, C. E., Stevenson, D., Coyle, M., Rees, R. M., Skiba, U. M., Sutton,

756   M. A., Cape, J. N., Dore, A. J., Vieno, M., Simpson, D., Zaehle, S., Stocker, B. D.,

757   Rinaldi, M., Facchini, M. C., Flechard, C. R., Nemitz, E., Twigg, M., Erisman, J.

758   W., Butterbach-Bahl, K. and Galloway, J. N. 2015. Effects of global change

759   during the 21st century on the nitrogen cycle. Atmospheric Chemistry and

760   Physics, 15, 13849-13893. DOI:10.5194/acp-15-13849-2015

761   Futter, M. N., Erlandsson, M. A., Butterfield, D., Whitehead, P. G., Oni, S. K., and Wade,

762   A. J. (2013). PERSiST: the precipitation, evapotranspiration and runoff

763   simulator for solute transport. Hydrol. Earth Syst. Sci. Discuss., 10, 8635-8681,

764   DOI:10.5194/hessd-10-8635-2013.

765   Goffe, P. 2013. The Nitrates Directive, Incompatible with Livestock Farming?. Policy

766   Paper No. 93, 30th May. Notre Europe – Jacques Delors Institute. Dowloable

767   from:

768   www.institutdelors.eu/media/nitratesdirective-legoffe-ne-jdi-may13.pdf

769   Hans J.M. Van Grinsven, H.J.M., Tiktaka, A., Rougoorb, C.W. 2016. Evaluation of the

770   Dutch implementation of the nitrates directive, the water framework directive

771   and the national emission ceilings directive. NJAS - Wageningen Journal of Life

772   Sciences, 78, 69–84. DOI: 10.1016/j.njas.2016.03.010

773   Haury, J., Peltre, M. C., Tremolieres, M., Barbe, J., Thiebaut, G., Bernez, I., Daniel, H.,

774   Chatenel, P., Haan-Archipof, G., Muller, S., Dutartre, A., Laplace-Treyture, C.,

775   Cazaubon, A., and Lambert-Servien, E. 2006. A new method to assess water

776 trophy and organic pollution-the Macrophyte Biological Index for Rivers

777 (IBMR): its application to different types of river and pollution. Hydrobiologia,

778 570, 153-158. DOI: 10.1007/s10750-006-0175-3

779 Howarth, R., Swaney, D., Billen, G., Garnier, J., Hong, B. G., Humborg, C., Johnes, P.,

780 Morth, C. M. and Marino, R. 2012. Nitrogen fluxes from the landscape are

781 controlled by net anthropogenic nitrogen inputs and by climate. Frontiers in

782 Ecology and the Environment, 10, 37-43. DOI:10.1890/100178

783 Howden, N. J. K., Burt, T. P., Worrall, F., Whelan, M. J. and Bieroza, M. 2010. Nitrate

784 concentrations and fluxes in the River Thames over 140 years (1868-2008): are

785 increases irreversible? Hydrological Processes, 24, 2657-2662. DOI:

786 10.1002/hyp.7835

787 Jackson, B. M., Browne, C. A., Butler, A. P., Peach, D., Wade, A. J. and Wheater, H. S.

788 2008. Nitrate transport in Chalk catchments: monitoring, modelling and policy

789 implications. Environmental Science & Policy, 11, 125-135.

790 DOI:10.1016/j.envsci.2007.10.006

791 Kleijn, D., Sutherland, W.J. 2003. How effective are European agri-environment

792 schemes in conserving and promoting biodiversity? Journal of Applied Ecology,

793 40, 947–969. DOI: 10.1111/j.1365-2664.2003.00868.x

794 Kotti, M.E., Vlessidis, A.G., Thanasoulias, N.C., and Evmiridis, N.P. (2005). Assessment

795 of river water quality in Northwestern Greece. Water Resources Management,

796 19(1), 77-94. DOI: 10.1007/s11269-005-0294-z

797 Kuik, O. 2006. Ex ante and ex post costs of implementing the Nitrates Directive: Case

798 study in the framework of the project 'Ex post estimates of costs to business

799 of EU environmental policies'. Commissioned by European Commission, DG

800   Environment, Unit G.1 Sustainable Development and Economic Analysis.

801   Framework contract No ENV.G.1/FRA/2004/0081. Downloable from:

802   ec.europa.eu/environment/enveco/ex_post/pdf/nitrates.pdf

803   Luo, G. J., Kiese, R., Wolf, B. and Butterbach-Bahl, K. 2013. Effects of soil temperature

804   and moisture on methane uptake and nitrous oxide emissions across three

805   different ecosystem types. Biogeosciences, 10, 3205-3219. DOI:10.5194/bg-

806   10-3205-2013

807   Macdonald, G. K., Bennett, E. M., Potter, P. A. and Ramankutty, N. 2011. Agronomic

808   phosphorus imbalances across the world's croplands. Proceedings of the

809   National Academy of Sciences of the United States of America, 108, 3086-

810   3091.DOI: 10.1073/pnas.1010808108

811   Manolaki, P., Tsakiri, E., and Papastergiadou, E. 2011. Inventory of aquatic and riparian

812   flora of Acheron and Louros rivers, and Zirou Lake in Western Greece.

813   Fresenius Environmental Bulletin, 20(4), 861-874.

814   Matzdorf, B., Lorenz, J. 2010. How cost-effective are result-oriented agri-

815   environmental measures?—An empirical analysis in Germany. Land Use Policy

816   27, 535–544. DOI:10.1016/j.landusepol.2009.07.011

817   Nakićenović, N., J. Alcamo, G. Davis, B. de Vries, J. Fenhann, S. Gaffin, K. Gregory, A.

818   Grübler, T.Y. Jung, T. Kram, E.L. La Rovere, L. Michaelis, S. Mori, T. Morita, W.

819   Pepper, H. Pitcher, L. Price, K. Raihi, A. Roehrl, H.-H. Rogner, A. Sankovski, M.

820   Schlesinger, P. Shukla, S. Smith, R. Swart, S. van Rooijen, N. Victor and Z. Dadi

821   (2010), "IPCC Special Report on Emissions Scenarios", Cambridge University

822   Press, Cambridge, UK.

823   Ovezikoglou, V., Ladakis, M., Dassenakis, M., & Skoullos, M. (2003). Nitrogen,

824   phosphorus and organic carbon in main rivers of the western Greece. Global

825   NEST Journal, 5(3), 147-156.

826   Randall, N.P., Donnison, L.M., Lewis, P.J., James, K.L. 2015. How effective are on‑farm

827   mitigation measures for delivering an improved water environment? A

828   systematic map. Environmental Evidence, 4:18. DOI: 10.1186/s13750-015-

829   0044-5

830   Saggar, S., Jha, N., Deslippe, J., Bolan, N.S., Luo, J., Giltrap, D.L., Kim, D.G., Zaman, M.

831   and Tillman, R,W. 2013. Denitrification and $N_2O:N_2$ production in temperate

832   grasslands: processes, measurements, modelling and mitigating negative

833   impacts. Science of the Total Environment, 465(1), 173-195. DOI:

834   10.1016/j.scitotenv.

835   Skoulikidis, N., Amaxidis, Y., Bertahas, I., Laschou, S. and Gritzalis, K. 2006. Analysis of

836   factors driving stream water composition and synthesis of management

837   tools—A case study on small/medium Greek catchments. Science of the Total

838   Environment, 362, 205-241. DOI:10.1016/j.scitotenv.2005.05.018

839   Skuras, D., Wade, A., Psaltopoulos, D. Whitehead, P. Kontolaimou, A. and Erlandsson,

840   M. 2014. An interdisciplinary modelling approach assessing the cost-

841   effectiveness of agri-environmental measures on reducing nutrient

842   concentration to WFD thresholds under climate change: the case of the Louros

843   catchment. Operational Research, 14(2), 205-224. DOI: 10.1007/s12351-014-

844   0158-5.

845   Söderholm, P. and Christiernsson, A. 2008. Policy effectiveness and acceptance in the

846   taxation of environmentally damaging chemical compounds. Environmental

847   Science and Policy, 11. 240-252. DOI:10.1016/j.envsci.2007.10.003

848   Verhoeven, J. T. A., Arheimer, B., Yin, C. Q. and Hefting, M. M. 2006. Regional and

849   global concerns over wetlands and water quality. Trends in Ecology &

850   Evolution, 21, 96-103. DOI: 10.1016/j.tree.2005.11.015

851   Wade, A.J., Durand, P., Beaujouan, V., Wessel, W.W., Raat, K.J., Whitehead, P.G.,

852   Butterfield, D., Rankinen, K. and Lepisto, A. 2002a. A nitrogen model for

853   European catchments: INCA, new model structure and equations, Hydrology

854   and Earth System Sciences, 6, 559-582. DOI:10.5194/hess-6-559-2002, 2002.

855   Wade, A. J., Whitehead, P. G. and Butterfield, D. 2002b. The Integrated Catchments

856   model of Phosphorus dynamics (INCA-P), a new approach for multiple source

857   assessment in heterogeneous river systems: model structure and equations,

858   Hydrology and Earth Systems Sciences, 6, 583-606.

859   Wang, HJ. and Wang, HZ. 2009. Mitigation of lake eutrophication: Loosen nitrogen

860   control and focus on phosphorus abatement. Progress in Natural Science,

861   19(10), 1445–1451. DOI:10.1016/j.pnsc.2009.03.009

862   Zerefos, C., Capros, P., Natsis, A. Papandreou, A., Sabethai, I and Yfantopoulos, I. (eds.)

863   2011. The environmental, economic and social impacts of climate change in
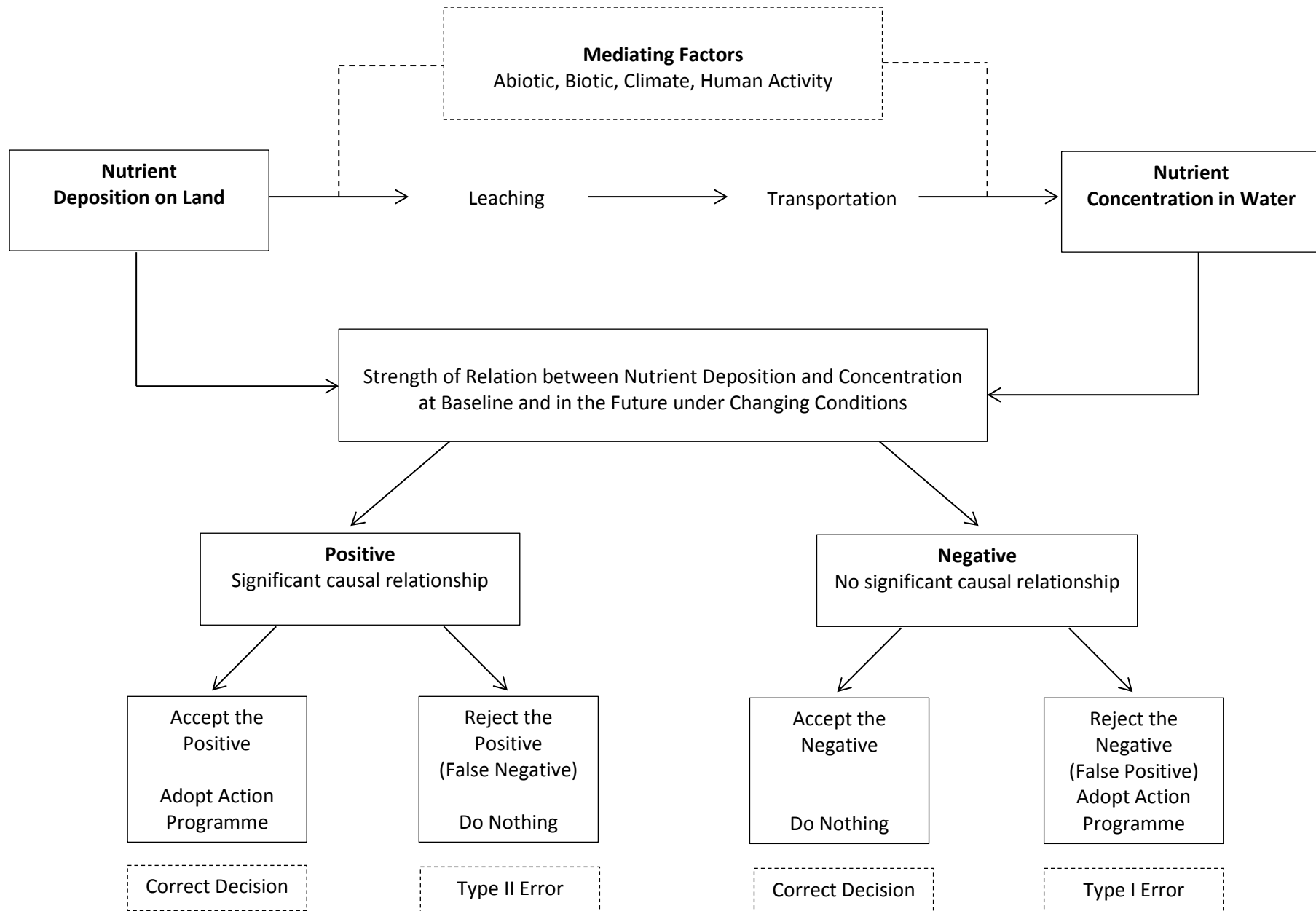
864   Greece. Bank of Greece (in Greek).

Figure 1. Sources of False Positive and Negative Errors in the design of agri-environmental measures.

A Decision Making Process for Rejecting False Positives and False Negatives in the Design and Implementation of Agri-environmental Policies

Socio-economic Models

Consultation - Participation Models

Scientific community:
- Published research
- Elite Interviews
- Structured Questionnaires

Stakeholders:
- Focus Groups
- Workshops

Decision Makers:
- Elite Interviews

Economic Models

Cost Effectiveness:
- Spreadsheet Techniques
- Optimization techniques
- Expert Based Scoring System
- Regression Approaches
- Bayesian Belief Networks

Science Models

Hydrology and Water Quality Models

Hydrology:
- PERSiST

Modelling Nutrients and sediments:
- INCA-N
- INCA-P
- HBV-N
- MIKE SHE
- AGNPS
- SWAT

Climate Change Models

Atmosphere-Ocean General Circulation Models:
- HadCM3
- ECHAM
- IPSL-CM4
- CCSM3

Land Use and Crop Production Models

Land Use
- LandSFACTS
- CLUE

Crop Production
- DSSAT
- CROPSYST
- AQUACROP

Step 1
- Recognize possible non-compliance issues
- Define the problem

Step 2
Baseline Model of:
- Nutrient and
- Sediment transportation

Step 3
- **Reject false positives and false negatives**
- Identify possible mitigation measures

Step 4
- Effectiveness of mitigation measure for nutrient reduction using dynamic models

Step 5
- Determine flow and quality compliance of simulated mitigation against thresholds

Step 6
- **Reject false positives and false negatives**
- Assess economic cost effectiveness

Step 7
- Determine base-line flow and quality for climate change and land use scenarios

Step 8
- **Reject false positives and false negatives**
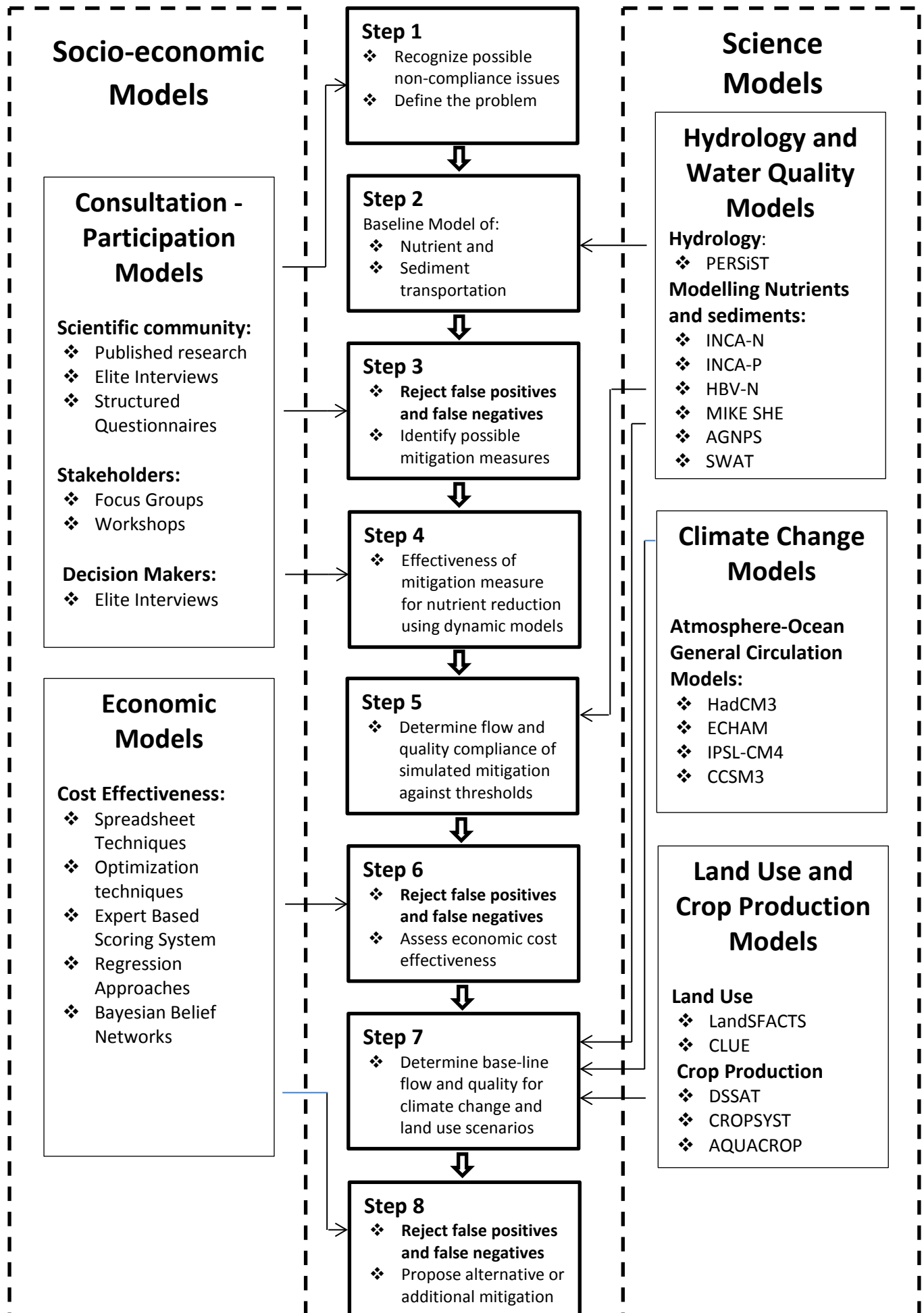- Propose alternative or additional mitigation

Figure 2. The eight step decision process integrating socio-economic and science models.

Baseline Input
1) Temperature and precipitation from 3 stations
2) Land cover from CORINE to estimate cultivated and natural land
3) Anthropogenic nutrient inputs from fertilizers and sewage
4) Biological fixation of N for non-cultivated land
5) Annual atmospheric values of dry and wet deposition
6) Hydrographic network, water chemistry and flow
7) Soil map for producing soil boxes

Calibration of
Hydrological Model
PERSiST

Calibration of
Integrated
Catchment Model
INCA-N

**Baseline
Simulation Model**

Are Baseline
Nutrient Loads
Above Threshold?

Yes

No

Adopt **Mitigation**
and Modify
Baseline Inputs

Modify Inputs for
Climate and **Land Use**
Change **Scenarios**

Re-estimate
Nutrient Loads from
Simulated **PERSiST**
and **INCA-N**

Re-estimate
Nutrient Loads from
Simulated **PERSiST**
and **INCA-N**

Are **Mitigation**
Nutrient Loads
Above Threshold?

Are **Scenario**
Nutrient Loads
Above Threshold?

Yes

No
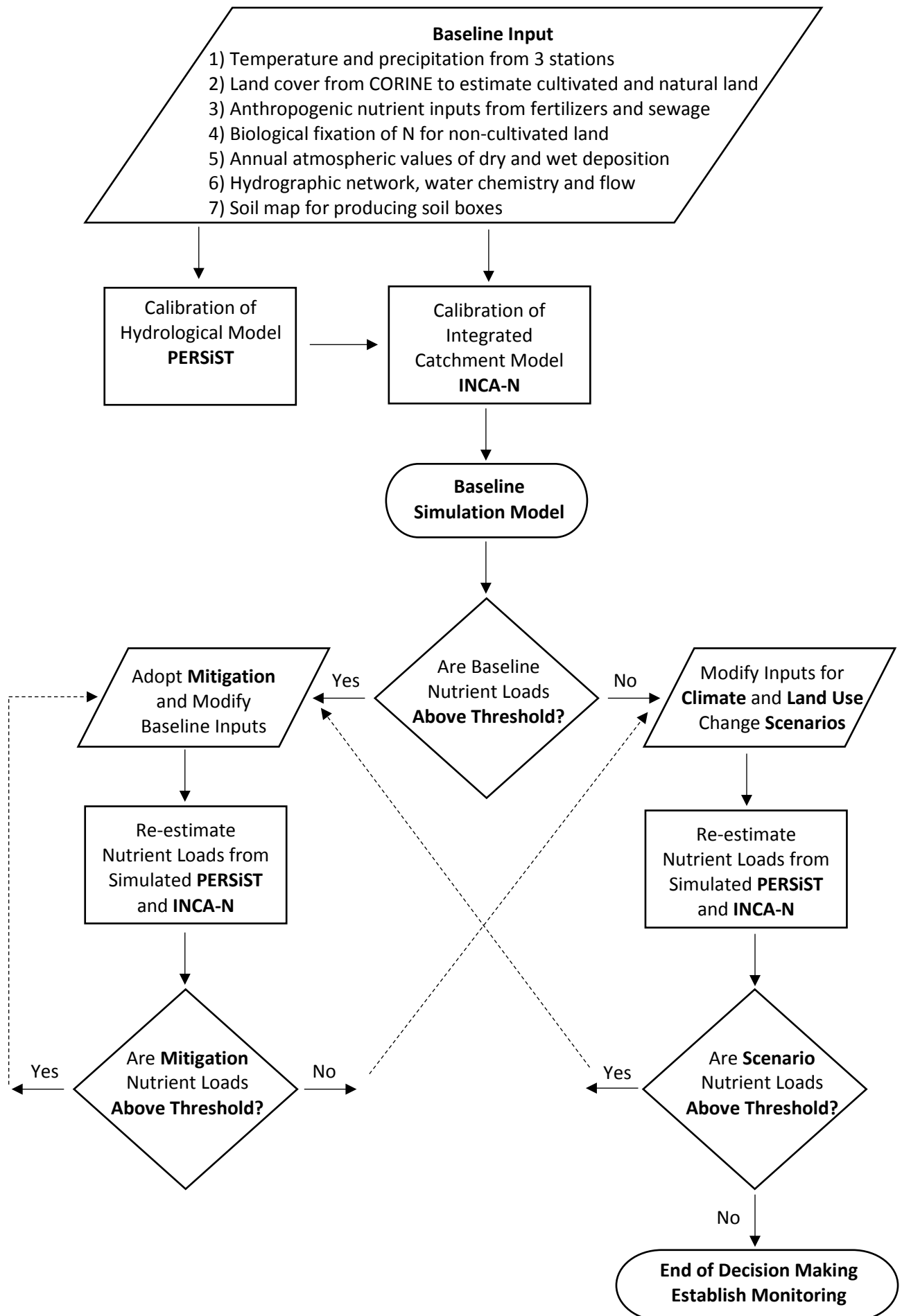
Yes

No

**End of Decision Making
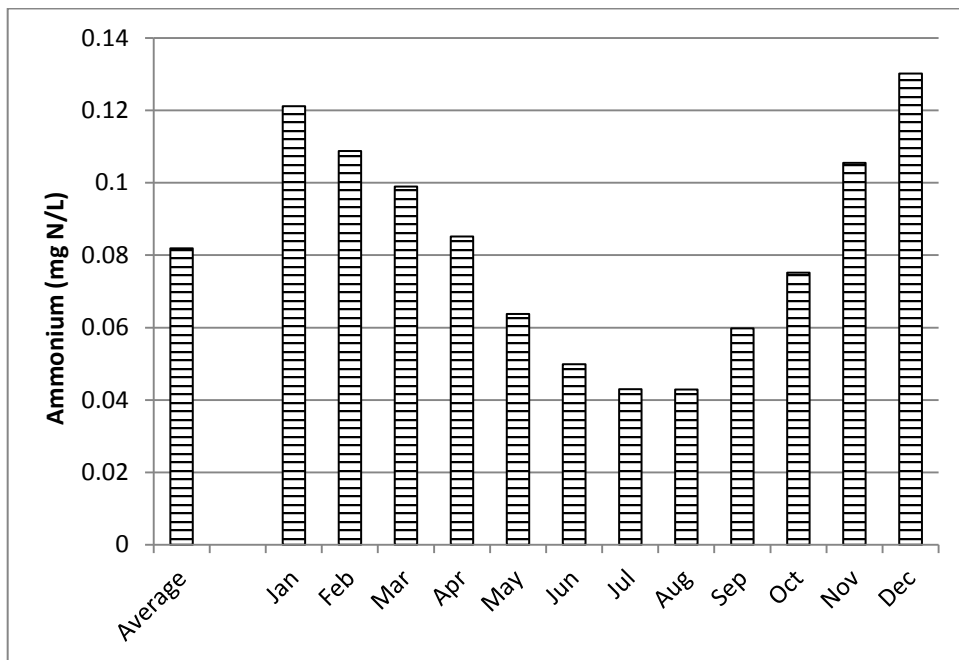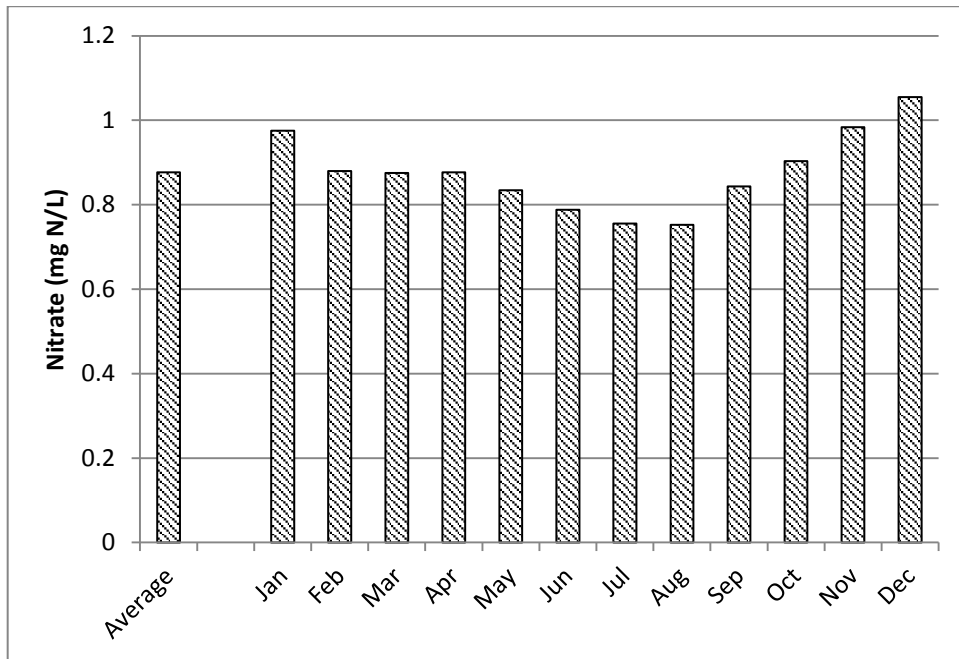Establish Monitoring**

Figure 3. A flowchart of the baseline modelling, mitigation and simulation scenarios for the Louros catchment.
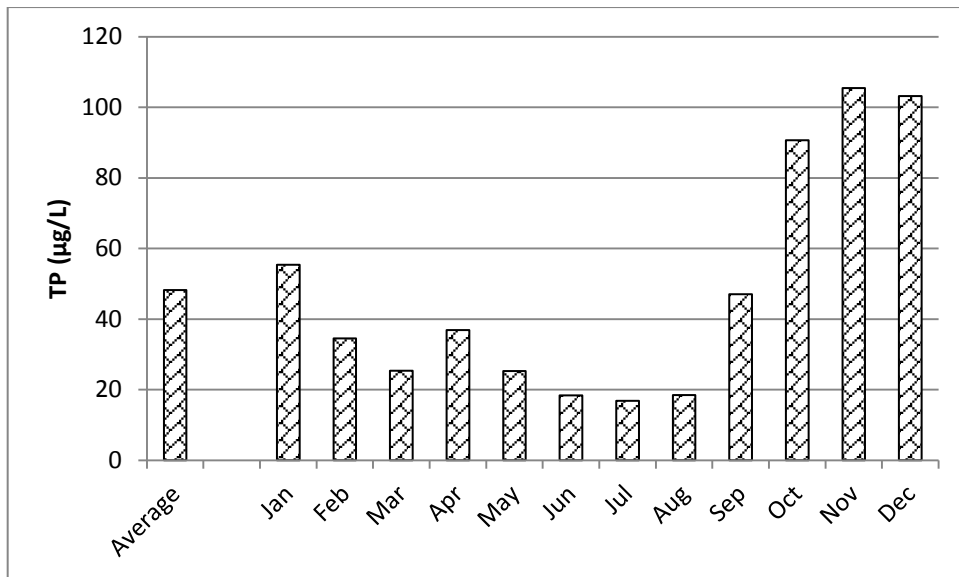
Figure 4. Baseline simulation of average and monthly nitrate, ammonium and Total Phosphorous (TP) concentrations.

Table 1. The emergence and indicative underlying reasons of false positive and negative decisions in the design and implementation of agro-environmental programs.

True deposition level from agriculture today	True impact of agriculture to concentration levels today	Policy outcome towards establishing an AE program	Type of Error	Indicative reasons of error
High	Low	Yes	False positive	• Poorly implemented baseline survey failing to detect the relatively low contribution of agriculture to pollution levels by ignoring other sources of pollution such as naturally occurring nutrient release, untreated sewage, leaks

				from poorly septic tanks, upstream fish farming, poorly treated industrial wastes, etc.
				• Failure to recognize the relatively low contribution of agriculture to pollution with respect to other activities due to lack of consultation with local stakeholders on the real condition of septic tanks, waste water treatment plans in the industry or the municipalities, illegal activities, etc.
				• Failure to model the deposition-transportation-concentration relationship which, due to abiotic or biotic reasons (existence of underground water reservoirs, soil conditions favouring high denitrification, deposition at river banks, high plant uptake, etc) results to the low contribution of agriculture to pollution
High	Low	No	None	
High	High	No	False negative	• Poorly implemented baseline survey failing to detect high deposition levels and attributing the observed high concentration levels to non-agricultural sources

				• Failure to recognize high deposition levels due to lack of consultation with local stakeholders on, for example, market price or policy transmitted incentives for the use of excess fertilization, or lack of information on widely adopted bad agricultural practices • Failure to model the direct deposition-transportation-concentration relationship
High	High	Yes	None	
Low	Low	Yes	False positive	• Poorly implemented baseline survey failing to detect low pollution levels and their source, e.g., water samples collected during peak concentration end of summer months of an unusually dry year • Failure to recognize the relatively low deposition rates and low contribution of agriculture to pollution due to lack of knowledge of e.g., specific, locally adapted low-input agro-systems • Failure to model the deposition-transportation-concentration relationship

Low	Low	No	None	
Low	High	No	False negative	• Poorly implemented baseline survey failing to detect that despite low deposition rates, very bad farming practices (time of application, irrigation methods, etc.) or excess water abstraction for non-agricultural uses may lead to high nutrient concentration in a possibly very unstable water ecosystem • Failure to recognize the relatively low deposition rates but high impact of agriculture to pollution due to lack of knowledge on e.g., local water management practices that caused irreversible interventions leading to low water circulation, low water oxygenation, high solar radiation, etc. • Failure to model the high impact of low deposition rates on concentration levels
Low	High	Yes	None	

Table 2. Simulated nutrient concentrations under the baseline and different climate change, land use change and combined scenarios.

	N-NO$_3$ (mg/l)	N-NH$_4$ (mg/l)	SRP (µg/l)	TP (µg/l)
Baseline	0.88	0.08	43.2	48.2
Agro-environmental Mitigation Measures				
Mitigation 1	0.86	0.08	31.4	36.5
Mitigation 2	0.86	0.08	30.0	35.1
Climate and Land Use Changes Scenarios				
Climate Change Best Scenario (KNMI+B1)	0.86	0.07	42.8	48.8
Climate Change Worse Scenario (Hadley+A2)	0.89	0.08	43.7	50.6
CAP Induced Land Use Change at 25%	0.86	0.08	31.4	36.5
CAP Induced Land Use Change at 30%	0.86	0.08	30.1	35.2
Scenarios from Combined Changes				
Mitigation 2 and Climate Change (KNMI+B1)	0.84	0.07	29.1	35.4
Mitigation 2 and Climate Change (Hadley+A2)	0.87	0.08	29.9	37.5
CAP Land Use Change at 30% and Climate Change (KNMI+B1)	0.84	0.07	29.4	35.1
CAP Land Use Change at 30% and Climate Change (Hadley +A2)	0.87	0.08	30.6	36.8

Source: Authors' estimates from INCA-N and INCA-P simulations.

Table 3. Estimates of the average mitigation cost in the Louros watershed, Greece.

A. Average estimates of fertilizer abatement under Mitigation 1								
	Area (ha)	Fertilizer Application		Reduced Fertilizer		Cost (€/ha)	Average Abatement Cost (€/Kg)	
		N-Kg/ha	P-Kg/ha	N-Kg/ha	P-Kg/ha		N	P
Cotton	337	110.0	50.0	64.6	29.4	657.2	10.2	22.4
Corn	3,306	240.0	200.0	117.0	97.5	521.6	4.5	5.4
Medic	4,009	80.0	100.0	35.0	43.8	437.2	12.5	10.0
Citrus	2,093	300.0	40.0	75.0	10.0	538.1	7.2	53.8
All	9,745	182.6	119.3	72.4	54.2	495.1	6.8	9.1
B. Average estimates of nutrient concentration abatement under Mitigation 1								
	Area (ha)	Simulated average concentration						

		at baseline		after mitigation		Cost of mitigation (€)	Average Abatement Cost [€/(µg/l)]	
		NO_3 (mg/l)	TP (µg/l)	NO_3(mg/l)	TP (µg/l)		NO_3	TP
All	9,745	0.88	48.2	0.86	36.5	4,825,061	301,566	412,398

Source: Authors' estimates from the RICA/FADN database, agronomic information and Focus Groups with stakeholders.