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Psaltopoulos, D., Wade, A. ORCID: https://orcid.org/0000-0002-5296-8350, Skuras, D., Kernan, M., Tyllianakis, E. and Erlandsson, M. (2017) False positive and false negative errors in the design and implementation of agri-environmental policies: a case study on water quality and agricultural nutrients. Science of the Total Environment, 575. pp. 1087-1099. ISSN 0048-9697 doi: 10.1016/j.scitotenv.2016.09.181 Available at https://centaur.reading.ac.uk/67069/

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

Publisher: Elsevier

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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

Demetrios Psaltopoulos [1], Andrew J. Wade[2], Dimitris Skuras [3], Martin Kernan[4], Emmanouil Tyllianakis [5] and Martin Erlandsson[6]

[1] Professor, Department of Economics, University of Patras, University Campus, Rio, 26504 Greece; email: skuras@econ.upatras.gr

[2] Professor, Department of Geography and Environmental Science, University of Reading, Whiteknights, , Reading, RG6 6DW, UK;

[3] Professor, Department of Economics, University of Patras, University Campus, Rio, 26504 Greece; email: dempsa@upatras.gr

[4] Senior Project Manager, CBEC Eco-Engineering UK Ltd, The Green House, Beechwood Park North, Inverness, IV2 3BL, UK; email: m.kernan@cbecoeng.co.uk

[5] Ph.D Candidate, Department of Economics, University of Patras, University Campus, Rio, 26504 Greece; email: etyllianakis@upatras.gr

[6] Research Fellow, Department of Earth Sciences, Uppsala University, Sweden; email: martin.erlandsson@geo.uu.se

Acknowledgement

This work was carried out as part of the REFRESH FP7 Project: Adaptive Strategies to Mitigate the Impacts of Climate Change on European Freshwater Ecosystems, funded by EU Seventh Framework Program (Grant Agreement 244121): www.refresh.ucl.ac.uk .

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

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

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

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

the Water Framework (WFD) and the Habitats and Birds Directives. Over the years, agri-environment policy has emerged as one of the most important elements of the Common Agricultural Policy (CAP) in terms of its budgetary size and the proportion of participating farmers and farmland.

The effectiveness of AEMs to enhance biodiversity (Batáry et al., 2015; Kleijn and Sutherland, 2003) and protect aquatic environments from agricultural pollution has been reviewed very extensively, has been questioned and criticized (Grinsven et al., 2016; Buckley et al., 2016; Matzdorf and Lorenz, 2010; Randall et al., 2015). The results are disparate mainly due to the plethora of applied measures, the heterogeneity in the application agroecosystems and their baseline status, the variability in set targets and the way these targets are monitored. Decision making for the adoption and establishment of AEMs targeting the reduction of nutrient concentration in water is implemented, very frequently, without a comprehensive and integrated plan. For example, AEM decision makers may be unable to control for non-agricultural nutrient contributing activities, industrial or municipal, which are beyond their institutional jurisdiction. As a result, AEM decision makers tend to set program targets on inputs (quantities of mineral fertilizers, manure or irrigation water) rather than on downstream chemical water quality or environmental status. Consequently, an AEM can be considered to be very effective because it managed to reduced inputs to the targeted level when, in reality, the AEM had marginal or no effect in reducing nutrient loads downstream.

In decision-making, a false positive, known in statistics as Type I error, refers to the situation where the presence of a condition is assumed when in reality there is not such a situation. A false negative, known in statistics as Type II error, refers to the

situation where no presence of a condition is assumed when in reality there is one. As such, the words “positive” and “negative” correspond to the answers “yes” or “no” to the question “is upstream agricultural activity responsible for downstream pollution?”. In this sense, a false positive coincides with “yes (positive) agriculture is responsible for downstream pollution” when in reality this is not true (false). Correspondingly, a false negative decision is committed when answering “no (negative) agriculture is not responsible for downstream pollution” when in reality it is responsible (false). In addition to the current situation, action programmes should consider whether the nutrients issue is likely to increase or decrease in the future. In this case the decision question “is upstream agricultural activity likely to become responsible for downstream pollution in the next 7-10 years?” can lead to false positives if action programmes address areas where the nutrients issue is neither currently nor in the future likely to become important. In this context, false negatives emerge when action programmes address areas where the nutrients issue is currently very important and may likely remain so in the future (false negative). In any case, an informative forecast of the future effects of agriculture on the environment can alert policy to be ready to establish programmes or to respond by modifying the incentives provided in existing programmes.

The aim of this paper is to propose an integrated decision making framework for designing and establishing AEMs targeting nutrient reduction. This decision making framework reduces the risk of committing false positives and wasting financial resources or the risk of committing false negatives and not protecting the environment. Section 2 of this work, briefly reviews the sources contributing to the 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

direct relationship between nutrient deposition on the field and downstream nutrient concentration. As such, it fails to take account of the static abiotic environment (geology and soil) and the dynamics of human activity and climate change, as well as of the changing and fluctuating water supply. Thus, we should not presume a linear and static relationship between deposition and concentration on which to build solid and robust AEM.

Figure 1 below attempts to sketch how false positives and false negatives may be generated in agri-environmental policy-making. The upper part of the diagram provides a coarse picture of the nutrient deposition-leaching-transportation-concentration process and how this process is influenced by abiotic, biotic, human activity and climate change factors. Under abiotic factors we refer to those physical processes pertinent to the geology, topography, soil physical and chemical properties. Under biotic factors, we refer to the whole range of sources that contribute to nutrient deposition such as land uses other than agriculture and animal activity other than livestock and/or grazing. Under human activity factors, we refer to agriculture and other activities contributing nutrients and including municipal and industrial sources coming from septic tanks or other devices of establishments that are not connected to municipal wastewater networks, animal wastes, food processing, etc. In addition, activities other than agriculture, may have an impact on the hydrology and especially on the quantity and frequency of water provided to water courses. Beyond irrigation and its corresponding drainage networks, examples include water extraction for municipal and industrial uses and sometimes small or large scale energy production from hydro electrical power plants. Finally, climate and especially temperature and

runoff are important factors determining nutrient cycling and transport (Howarth et al., 2012).

The relationship between agricultural inputs and instream nutrient concentrations is not a simple one, for example because of a large groundwater store that may act as nutrient reservoir or because water flows may change, or plant uptake may increase, or land use may change (Jackson et al., 2008; Howden et al., 2010). Thus, the underlying relationship between nutrient deposition by agriculture and its impact on nutrient concentration in downstream water bodies may be important (yes-positive) or not (no-negative). However, without an integrated approach modelling the relationship between nutrient input and instream concentrations there will be uncertainty as to whether the policy can address the input-output relationship accurately, and therefore avoid the risk of false positives and false negatives. Table 1 provides an indicative list of false positive or negative decisions along with connotative reasons causing these deceptive decisions. A similar table may be generated if dynamic changes caused by land use and climate change are taken into account. In this context, dynamics may generate an agricultural pollution issue in areas that currently have not such an issue and vice versa.

In this work we focus on two broad areas within the policy design process which may contribute to the generation of false positives and false negatives:

- Appropriate baseline monitoring and modelling of the nutrient deposition-concentration function and the resultant baseline abatement function measured in terms of nutrient concentration in the water downstream and,

- Forecasting and incorporation of changes resultant from human activity and climate change and the resultant dynamic abatement function again measured in terms of nutrient concentration in the water downstream

Taking into account the long-term horizon for implementing an agri-environment programme, policy design, and especially baseline modelling, should consider dynamic changes that may considerably alter the initial conditions that lead to the adoption or the rejection of an agri-environment programme in a specific area. For example, within a seven year agri-environment planning horizon, several changes may occur in land use, in agricultural production or/and even climatic conditions. Land use changes may be instigated by agricultural policy changes such as the CAP, which may lead to the abandonment of agricultural production or to the drastic change in the adoption of cultivations with different nutrient applications (Barbayiannis et al., 2011). One vivid example is the decoupling of Pillar 1 subsidies, which in some EU areas, has induced the abandonment of several cultivations or the shift to other crops, including nitrogen fixing legumes and the consequent reduction in nitrogen deposition. At the same time other, economy-wide developments, may affect (increase) agricultural input prices resulting to a rationalization and the consequent reduction of nutrient deposition.

In conventional policy design, targets are set on deposition, assuming that a proportional reduction will be achieved in the corresponding concentration of nutrients. The Nitrates Directive and several other EU, national and regional policies set such targets. This approach promotes “one-size fits all” policy and fails to take account of the aforementioned specificities of the environment and of human activity in the target-area(s), that call for a case-specific and “tailor made” approach to agri-

environmental target setting. In this work it is suggested that the baseline situation should be modelled according to an integrated framework accounting for dynamic changes. In this respect we minimize the risk of false positives and false negatives. To this end we advocate a procedure that uses a dynamic, mass-balance water quality model to help explain the input (deposition) – output relationship and integrates science and socio-economic models to protect policy design from committing false positives or false negatives (Skuras et al., 2014).

Figure 2 depicts this approach in a sequence of policy design steps supported by science and social science methods and models. Once the non-compliance issue is recognized and defined (step 1) with the support of existing data and socio-economic public participation models, an integrated model of nutrient and sediment transportation within the catchment is proposed to be constructed (step 2). This step is supported by scientific models of nutrients and/or sediment transport that calibrate a baseline situation based on flow and hydrochemistry conditions of the catchment depicted by meteorological, soil-geological, flow, land use and water quality data. In step three, decision makers will have the capacity to avoid false negatives and false positives. False positives are usually generated by failing to take into account the whole range of sources contributing nutrients to the watershed and overestimate the contribution and impact of agriculture. In this context, adopting a policy to control nitrogen deposition from agriculture will not have an effect. Potentially, false positives may be generated by situations in which high nutrient deposition fails to show up in water nutrient concentrations for various reason including geology, e.g., extensive carstic phenomena that redirect nutrient rich water to neighbouring watersheds or to

underground water reservoirs, soil conditions that favour high denitrification, deposition at river banks, etc.

In steps 4 and 5, mitigation measures are proposed and their effect is examined according to the calibrated baseline model. This will allow the examination of the simulated effectiveness of the mitigation measures and hence, the prevention of false positives, by adopting measures that will not be effective or the prevention of false negatives, by rejecting measures that will be effective (step 6). In step 7 the baseline condition and the mitigation measures are re-estimated and simulated against changing conditions including climate, land use and production. This will allow the prevention of false positives in the sense that a deposition-concentration situation that seems positive today may be most likely ameliorated in the near future due to changing conditions, without the need of mitigation measures and thus, adopting a programme would be less appropriate (step 8). The same step will allow the prevention of false negatives in the sense that a seemingly unrelated deposition-concentration situation today may be most likely aggravated in the near future due to changing conditions and adopting mitigation measures under an agri-environment programme would be appropriate.

3. Case Study and Methods

3.1. The case study area of the Louros watershed

The Louros water catchment (926 km^2) is situated in the central-southern part of the Epirus (NUTS 2 region) Water District in Greece. The river rises in the mountains adjacent to the “Dodoni Oracle”, one of the most important and famous oracles of 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

[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

cultivation and per hectare. Then, they were asked to adapt them (increase or decrease) for the local corresponding cultivations and provide the reasons why they suggested adaptations. For example, maize cultivations, depending on the site and soil properties, accept during starter band fertilization an average of 1,000 kg of fertiliser with a NPK ratio of 16-20-0 were applied per hectare corresponding to 160 Kg of N, or 1,000 Kg of 18-12-8 fertiliser corresponding to 180 Kg of N per hectare. During surface fertilization the same plots accept usually 300 to 350 Kg of 25-0-0 fertiliser per hectare corresponding to 75 to 90 Kg of N. The corresponding phosphorous fertilization is about 120-200 Kg per hectare depending on soil needs by using either 18-12-18 or 16-20-0 fertiliser during starter band fertilization. Application of phosphorous during surface fertilization is rare in the case of maize.

Water chemical analyses carried out by the Greek Ministry of Rural Development and Food (MRDF) showed that at locations close to the estuary, the concentration of nutrients (nitrates, ammonium and total phosphorous) was relatively high at least during autumn and early winter. Following this rather weak evidence, in 2006 MRDF established the plains of the Louros catchment and part of the adjacent Arachthos catchment, as a Nitrification Vulnerable Zone (NVZ) under the Nitrates Directive. The plan that, as yet, has not been implemented due to Greece's financial crisis and consequent budgetary constraints, allows compensation for farmers of the aforementioned cultivations if they comply with measures or combination of measures including the set aside of land, maximum allowable fertilization levels and irrigation per cultivation.

283 *3.2. Baseline Modelling*

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

box, one soil box and one groundwater box. Each box is characterized by nine different parameters, which are specific for each land class. There are nine additional land cover specific parameters, related to properties such as snow melt, evapotranspiration and base flow index. PERSiST was calibrated against the observed flow for the period Jan 2001 – Sep 2012. The INCA-N parameters calibrated included soil denitrification, soil nitrification, soil mineralisation, plant NO_3 uptake, plant NH_4 uptake, in-stream nitrification, in-stream denitrification, initial groundwater nitrate and initial groundwater ammonium. The size of the point source (effluent concentration of ammonium) was also calibrated, the hydrological parameters of groundwater residence time was adjusted to improve the fit for base-flow conditions, and the drought runoff fraction was adjusted to keep more nitrogen in the soil during the dry summer months. The INCA-P parameters calibrated included soil phosphorus terms (Freundlich isotherm, weathering factor, sorption coefficient and equilibrium phosphorus concentrations), plant uptake, process rates response to temperature, immobilisation, initial labile and inactive soil P, reach ecology parameters for macrophytes and epiphytes, and groundwater phosphorus terms. Both models, INCA-N and INCA-P were calibrated against nutrient concentration data from monitoring stations operated by the Ministry of Environment and Energy with reasonably good overall results and goodness of fit measures.

3.3. Simulating mitigation measures and future changes

In this section we detail the processes for simulating the effects of mitigation measures and the effects of future climate and land use changes on the baseline. In 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

4.2 Agri-environmental Measures and Future Changes

The simulated effects of agri-environmental measures, climate change, land use change and their combinations on nutrient concentration at the reach nearest to the estuaries of Louros river are presented for the simulated concentrations of nitrate, ammonium, SRP and TP in Table 2. If the mitigation measures described in section 3 of this work are adopted, the simulated reduction in average nitrate and ammonium concentrations is not significant. Quantitatively, exactly the same changes can be effected by short-term land use changes induced by the CAP without any mitigation measure. Furthermore, as concerns long-term climate change impacts, even the effect of the worse scenario does not show any important impact on nitrates and ammonium concentrations. The imperceptible modelled net change in nitrate concentrations for the climate scenarios is due to the fact that the amount of nitrogen leaching from the soils decrease by approximately the same rates as the runoff. At the same time, the average discharge decreases by between 15 and 28 %, the amount of nitrogen leaching from the soils decreases by 15 % (KNMI and SMHI models) to 25 % (Hadley).

The amount of nitrogen transported to the estuaries is however substantially reduced, by 16.3 % for the KNMI climate, 17.0 % for the SMHI climate, and by 26.5 % for the Hadley climate. The main reason for the simulated decline in nitrate leaching is that longer water residence time in the soil and stream and less runoff meant that more of the nutrients were available for plant uptake which balances the additional fertiliser load under increased CO_2 concentrations. Furthermore, due to lower atmospheric deposition, the external loads were around 5 % lower for the climate 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

river's water quality as concerns nitrates to "Good" and comply with WFD's requirements, will be yet another false positive decision in the design of agri-environmental policy. A false negative may emerge if decision makers fail to recognize the effect of short-term land use changes at least on phosphorous. Climate change does not seem to exert an important positive (best scenario) or negative (worse scenario) effect. In all scenarios, TP is reduced, even slightly. Recent evidence shows that phosphorus can determine river phytoplankton growth irrespective of the nitrogen concentration (Wang and Wang, 2009) and the physical conditions of light, water temperature and residence time are important in lowland river catchments.

4.3. The Cost of a False Positive

For each one of the four major cultivations in the watershed the cost of the mitigation measures was estimated. In order to proceed in our calculations we carried out two focus groups with stakeholders and elite interviews with agronomists in the area. Farmers' income from the different cultivations was estimated from the Standard Gross Margins derived by the FADN database for the region of Epirus where the Louros watershed is situated. From the FADN database we also calculated initial estimates of the cost of fertilization, and the cost of cultivating lentils, as well as the SGM of the lentil for fodder. Elite interviews with agronomists were utilised to estimate the loss in production due to reduced fertilization and irrigation. Consequently, stakeholders were presented with the initial estimates during a focus group with the aim to discuss and adapt initial estimates of the exact effects of reduced fertilization and irrigation on production and on farmer's income. In the 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

not reduce nitrate concentration by more than 0.02 mg/l and TP concentration by 0.01 mg/l. These are negligible achievements at a highly disproportional cost.

To summarize the discussion so far, it can be argued that the abatement cost of agri-environmental programmes aiming to manage nutrient loads should be measured as a change in nutrient concentration and not at levels of abated fertilizer. In other words, the targets of such agri-environmental programmes and policies should be set at nutrient concentration levels and not at quantities of abated substance either in mineral fertilizer or in manure and slurry. This can be attained if, during the design of agri-environmental programmes, the status quo (baseline), the impacts of the mitigation measures and the impacts from likely future changes are simulated. Then, false positives and false negatives can be avoided, the cost-effectiveness of mitigation measures can be assessed and an appropriate monitoring system can be set up.

5. Conclusions

The EU's agri-environmental policy is a response to the growing public concern over the environmental impacts of agriculture. As such, agri-environmental policy attempts to meet requirements from the WFD, the Nitrates Directive and the Habitats Directive, the cornerstone of environmental conservation in Europe. Agri-environmental policy has grown to a tremendous budget (€36.6 billion spent in the 2007-2013 programming period across the EU) and power by affecting almost a quarter of the EU's utilized agricultural area. This work concerned only with programmes managing nutrient loads in freshwaters and not with other forms of agri-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

722 **Acknowledgments**

723 This work was carried out as part of the REFRESH FP7 Project: Adaptive Strategies to
724 Mitigate the Impacts of Climate Change on European Freshwater Ecosystems, funded
725 by EU Seventh Framework Program (Grant Agreement
726 244121): www.refresh.ucl.ac.uk. The authors would like to thank two anonymous
727 referees of this journal for their constructive comments, and the associate editor
728 professor Simon Pollard for his efforts and support.

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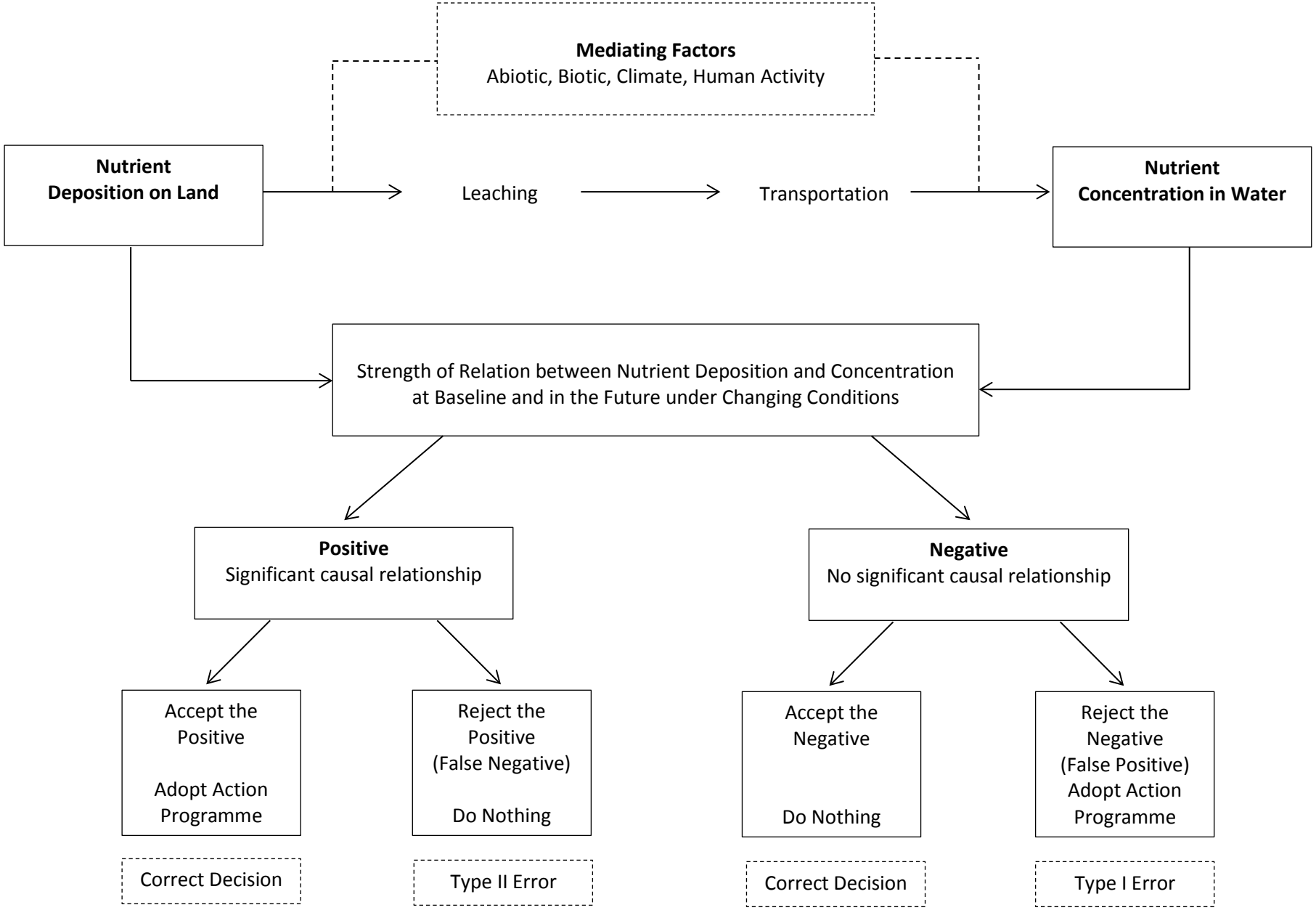

Mediating Factors
Abiotic, Biotic, Climate, Human Activity
Nutrient Deposition on Land
Leaching
Transportation
Nutrient Concentration in Water
Strength of Relation between Nutrient Deposition and Concentration at Baseline and in the Future under Changing Conditions
Positive
Significant causal relationship
Negative
No significant causal relationship
Accept the Positive
Adopt Action Programme
Reject the Positive
(False Negative)
Do Nothing
Accept the Negative
Do Nothing
Reject the Negative
(False Positive)
Adopt Action Programme
Correct Decision
Type II Error
Correct Decision
Type I Error

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

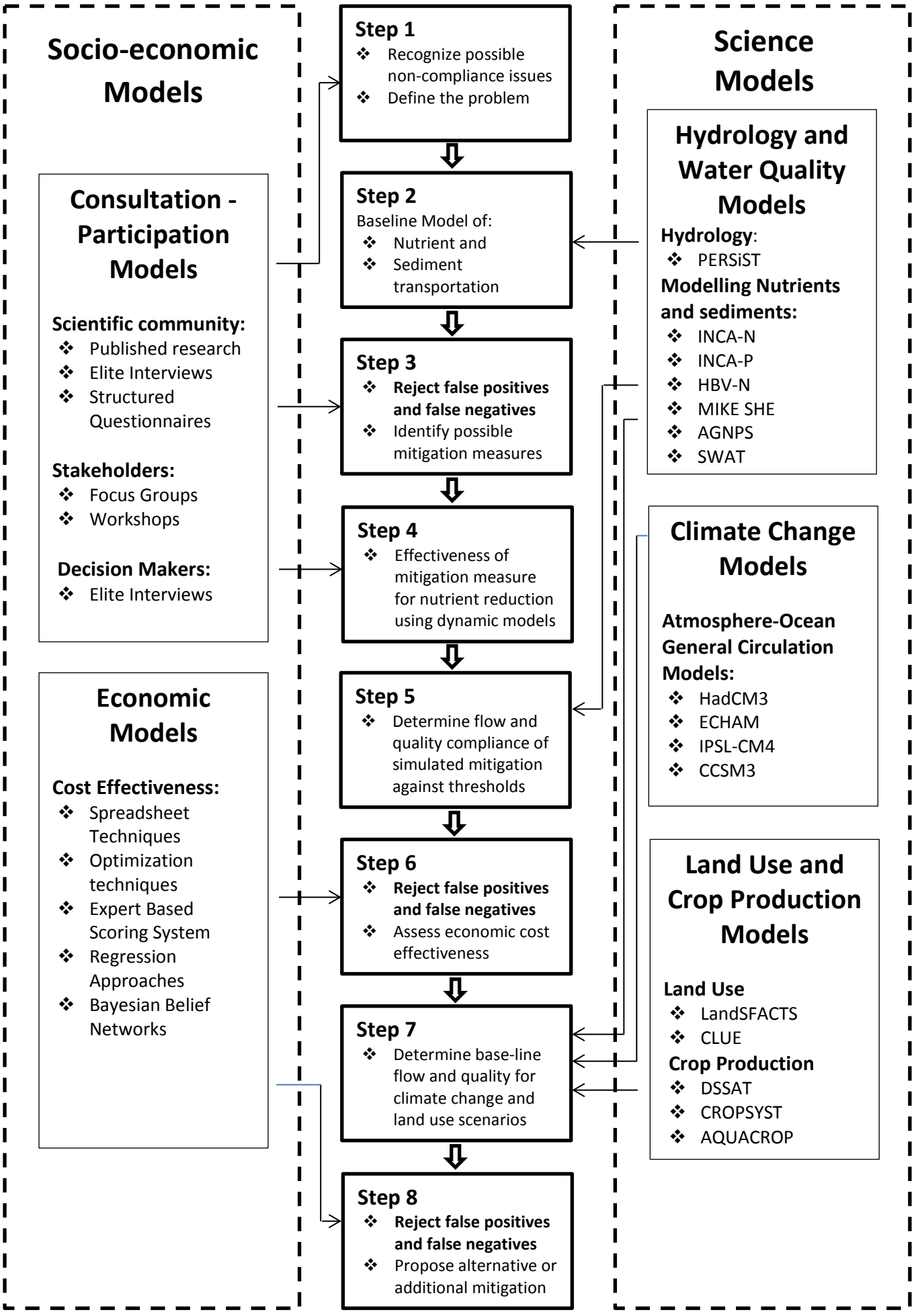


Figure 2. The eight step decision process integrating socio-economic and science models.

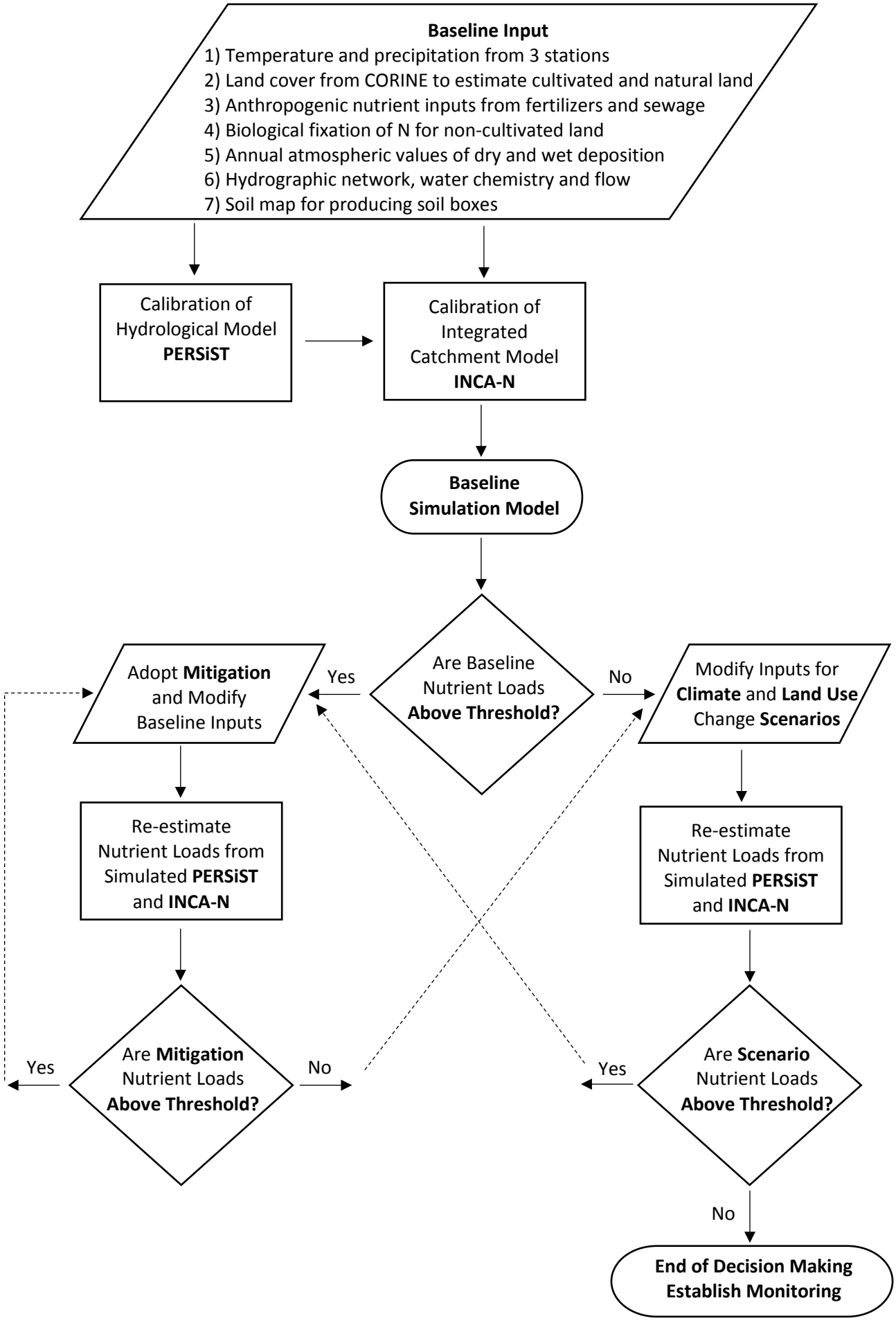
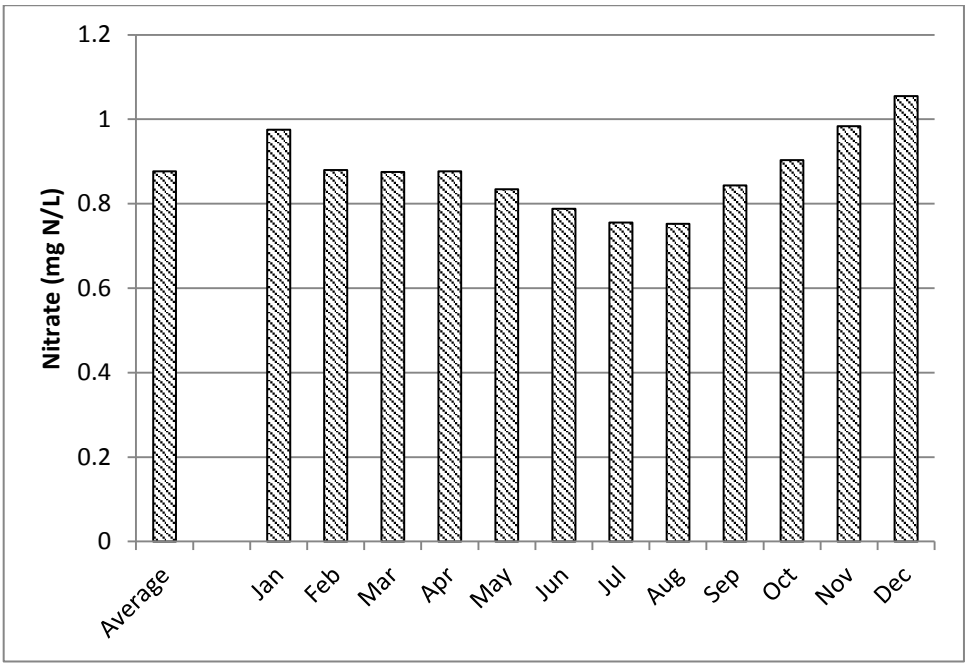
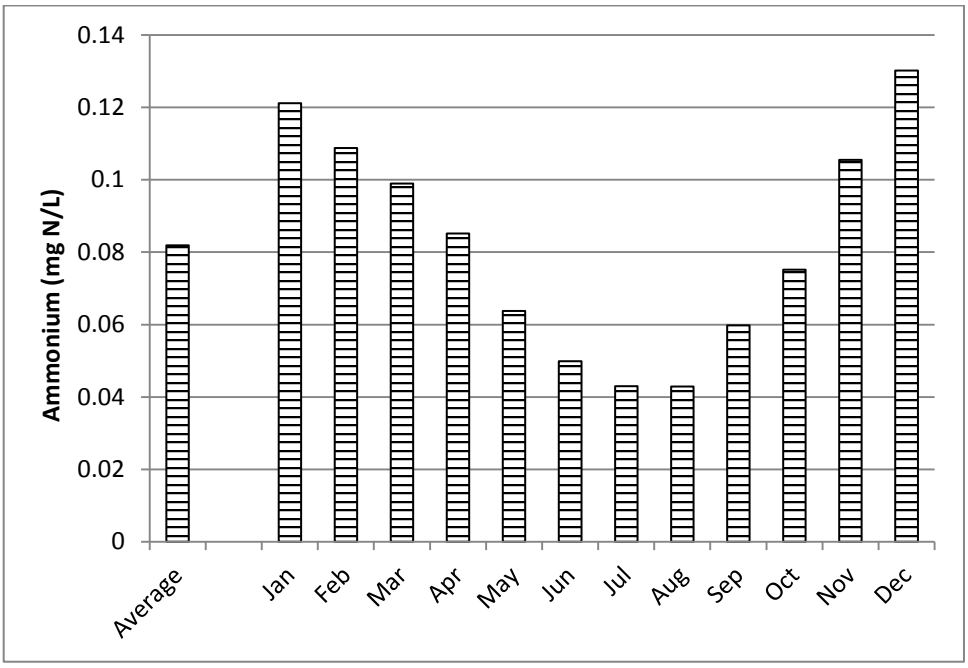

Baseline Input
1) Temperature and precipitation from 3 stations
2) Land cover from CORINE to estimate cultivated and natural land
3) Anthropogenic nutrient inputs from fertilizers and sewage
4) Biological fixation of N for non-cultivated land
5) Annual atmospheric values of dry and wet deposition
6) 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.


Nitrate (mg N/L)
1.2
1
0.8
0.6
0.4
0.2
0
Average
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec


Ammonium (mg N/L)
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
Average
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec

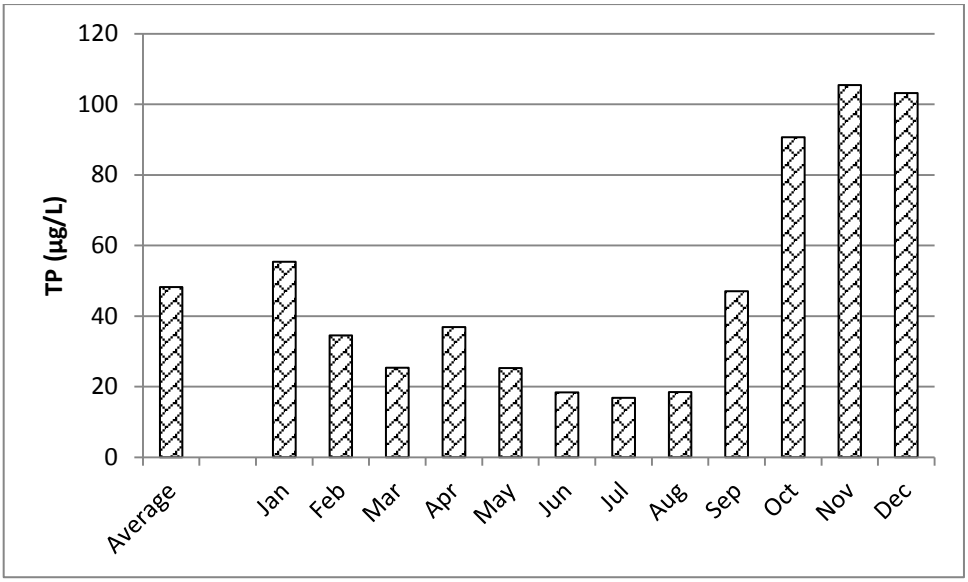


Figure 4. Baseline simulation of average and monthly nitrate, ammonium and Total Phosphorous (TP) concentrations.

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.

	$N\text{-}NO_3$ (mg/l)	$N\text{-}NH_4$ (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.