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1 **Using long time series of agricultural-derived nitrates for estimating catchment transit**2 **times**

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15
16 **Abstract:**

17 The estimation of water and solute transit times in catchments is crucial for predicting the response
18 of hydrosystems to external forcings (climatic or anthropogenic). The hydrogeochemical signatures of
19 tracers (either natural or anthropogenic) in streams have been widely used to estimate transit times
20 in catchments as they integrate the various processes at stake. However, most of these tracers are
21 well suited for catchments with mean transit times lower than about 4-5 years. Since the second half
22 of the 20th century, the intensification of agriculture led to a general increase of the nitrogen load in
23 rivers. As nitrate is mainly transported by groundwater in agricultural catchments, this signal can be
24 used to estimate transit times greater than several years, even if nitrate is not a conservative tracer.
25 Conceptual hydrological models can be used to estimate catchment transit times provided their

1 consistency is demonstrated, based on their ability to simulate the stream chemical signatures at
2 various time scales and catchment internal processes such as N storage in groundwater.

3 The objective of this study was to assess if a conceptual lumped model was able to simulate the
4 observed patterns of nitrogen concentration, at various time scales, from seasonal to pluriannual and
5 thus if it was relevant to estimate the nitrogen transit times in headwater catchments. A conceptual
6 lumped model, representing shallow groundwater flow as two parallel linear stores with double
7 porosity, and riparian processes by a constant nitrogen removal function, was applied on two paired
8 agricultural catchments which belong to the Research Observatory ORE AgrHys. The Global
9 Likelihood Uncertainty Estimation (GLUE) approach was used to estimate parameter values and
10 uncertainties. The model performance was assessed on (i) its ability to simulate the contrasted
11 patterns of stream flow and stream nitrate concentrations at seasonal and inter-annual time scales,
12 (ii) its ability to simulate the patterns observed in groundwater at the same temporal scales, and (iii)
13 the consistency of long-term simulations using the calibrated model and the general pattern of the
14 nitrate concentration increase in the region since the beginning of the intensification of agriculture in
15 the 1960s. The simulated nitrate transit times were found more sensitive to climate variability than
16 to parameter uncertainty, and average values were found to be consistent with results from others
17 studies in the same region involving modeling and groundwater dating.

18 This study shows that a simple model can be used to simulate the main dynamics of nitrogen in an
19 intensively polluted catchment and then be used to estimate the transit times of these pollutants in
20 the system which is crucial to guide mitigation plans design and assessment.

21
22 **Keywords:** lumped modeling, GLUE, surface water, shallow groundwater, hydro-chemical signatures,
23 response times.

24

25 **1. Introduction**

26

1 The estimation of water and solute transit times in catchments is crucial for predicting the response
2 of hydrosystems to external forcings (climatic or anthropogenic). As the hydrological and
3 geochemical signatures of streams represent the integration of the various processes at stake
4 (Aubert et al., 2013; Kirchner et al., 2001), they have been widely used to study hydrological
5 processes and to estimate transit times in catchments. For example, many studies compute transit
6 time distribution (TTD) by matching input (precipitation) and output (stream) time series of
7 concentrations of natural tracers like stable isotopes ^{18}O and ^2H or chloride (McGuire et al., 2007).
8 This method is well suited for catchments with mean transit times (MTT) lower than about 4-5 years
9 (Hrachowitz et al., 2009; McDonnell et al., 2010), otherwise it can lead to large MTT underestimation
10 (Stewart et al., 2010). However, it is more and more acknowledged that groundwater is playing a
11 major role in nutrient transport in agricultural catchments leading to MTTs exceeding several years
12 or decades, e.g. (Basu et al., 2012; Capell et al., 2012; Stewart et al., 2010). Several tracers of
13 anthropogenic origin like ^3H or CFCs can be used for estimating groundwater transit times spanning
14 several decades, but the current decline in their concentration in the atmosphere increases the
15 estimation uncertainty (Aquilina et al., 2012; Molénat et al., 2013; Stewart et al., 2010).

16 In many parts of the world, the intensification of agriculture led to dramatic increase of reactive
17 nitrogen input to agricultural watersheds in the second half of the 20th century (Galloway et al.,
18 2004). Agricultural inputs are known to be the main source of nitrogen in agricultural catchments
19 (Dunn et al., 2012; Howden et al., 2011b; Wang et al., 2013; Worrall et al., 2012). This led to a
20 general increase of the nitrogen load (mainly as nitrate) in many rivers of the world (Seitzinger et al.,
21 2010, Green et al., 2014). However, using nitrogen for estimating catchment transit time is not
22 straightforward as it is not a conservative tracer. Many studies on the impact of agriculture on
23 stream nitrate concentration have enlightened the lack of correlation between estimated N inputs
24 and outputs, but while part of them attribute this discrepancy mainly to N biotransformation,
25 highlighting the attenuation potential of the system (Montreuil et al., 2010; Billen and Garnier, 1999),
26 others suggest that it is mainly due to nitrate storage in the hydrosystem (vadose zone and

1 groundwater) leading to response times exceeding several years even in very small catchments (Basu
2 et al., 2012; Molenat et al., 2008; Owens et al., 2008; Ruiz et al., 2002a; Schilling and Spooner, 2006;
3 Tomer and Burkart, 2003; Wriedt and Rode, 2006).

4 During the last 30 years, more and more complex hydrological models have been developed
5 intending to simulate as accurately as possible the transformations and transfer of nitrogen in
6 catchments, while accounting as much as possible for the spatial heterogeneity of the processes.
7 However in models either semi-distributed, as INCA (Wade et al., 2002; Wade et al., 2001), SWAT
8 (Arnold et al., 1998); or fully-distributed, as TNT2 (Beaujouan et al., 2001; Moreau et al., 2013);,
9 MODFLOW-MT3D (McDonald and Harbaugh, 2003; Zheng et al., 2012) NitroScape (Duretz et al.,
10 2011), the complexity brings a number of issues related to overparameterization, parameter
11 uncertainty, and equifinality (Beven, 2006; Beven and Binley, 1992; Jakeman and Hornberger, 1993;
12 Perrin et al., 2001). As a consequence, there is renewed interest for lumped conceptual models, as
13 they are more parsimonious. However, the question remains if, while ignoring complexity and
14 heterogeneity, such models are still able to mimic realistic catchment processes and to provide “the
15 right answers for the right reasons” (Hrachowitz et al., 2013; Kirchner, 2006; Seibert and McDonnell,
16 2002).

17 A way to improve the realism of conceptual hydrological models is to make a better use of the
18 available information while assessing model performances by not relying on stream data only but
19 adding for example groundwater storage and chemical signature (Gupta et al., 2008; Seibert and
20 McDonnell, 2002), although getting acceptable simulation on both stream flow and groundwater
21 dynamics is not easy, even when considering only water (Fenicia et al., 2008; Gascuel-Odoux et al.,
22 2010b; Molénat et al., 2005). Similarly, using the temporal variations of the hydro-chemical signature
23 of streams can help in designing better model structures (Hrachowitz et al., 2011; Woodward et al.,
24 2013).

25 In temperate climates, nitrate concentrations in streams are characterized by different time scales of
26 variability. At the sub-daily scale, large variations can occur, which are linked either to hydrological

1 events (storm events) or to diurnal biological processes. As groundwater is the main pathway for
2 nitrate in agricultural catchments (Basu et al., 2012; Howden et al., 2011b; Molenat et al., 2008;
3 Woodward et al., 2013) these variations are not likely to affect significantly the nitrate mean transit
4 time. At the annual scale, seasonal cycles of nitrate concentrations are commonly described, either
5 positively or negatively correlated with flow (Martin et al., 2006 ; Betton et al., 1991) and attributed
6 mainly either to water mixing or to biological processes (e.g. Grimaldi et al., 2004; Martin et al., 2006;
7 Woodward et al., 2013). Inter-annual variability reflects the combined influence of year-to-year
8 variations of agricultural inputs and climate, the later determining the amount of water recharging
9 the groundwater which impacts its transit times (Gascuel-Oudou et al., 2010a; Heidebüchel et al.,
10 2013; Hrachowitz et al., 2009). On the long term (several decades), trends in nitrate concentration
11 reflect the impact of agricultural system evolution. However, studies are generally focused on only
12 one part of these temporal scales of variability. For example (Howden et al., 2011b) develop a model
13 based on the long time series (140 years) of nitrate concentrations on the Thames River. The model is
14 assessed on its ability to reproduce the dynamics of average yearly concentrations but without
15 looking at the seasonal variations. Green et al. (2014) used simple models for simulated annual
16 average nitrate concentrations over 10 catchments in USA and over the period 1964-2012, and
17 estimated from the models the mean travel time of nitrogen in the catchments ranging between 0
18 and 19 years depending on the model and the catchment. However they did not consider seasonal
19 pattern on the concentration too. Conversely, modelling seasonal variations of solute concentrations
20 in streams is usually carried out using only one (Benettin et al., 2013) or a few (Woodward et al.,
21 2013) water years.

22
23 The objective of this study was to assess if a conceptual lumped model was able to simulate the
24 observed patterns of nitrogen concentration at various time scales from seasonal to pluriannual and
25 thus if such a model was relevant to estimate the nitrogen transit time in headwater catchments.
26 This was achieved using a 20 year time-series of stream base flow and nitrogen concentration on two

1 paired agricultural catchments where a comprehensive survey of agriculture practices was available
2 over the same period. The conceptual lumped model used (ETNA modified from (Ruiz et al., 2002b))
3 represents shallow groundwater flow as two parallel linear stores with double porosity, and riparian
4 processes are represented by a constant nitrogen removal function. Parameter values and
5 uncertainties were estimated using the Global Likelihood Uncertainty Estimation (GLUE) approach
6 (Beven and Binley, 1992). The output variables of the model (water and nitrate output fluxes) were
7 used to compute the nitrate residence times in both catchments. The model was assessed according
8 to (i) its ability to simulate the contrasted patterns of stream flow and stream nitrate concentrations
9 at seasonal and inter-annual time scales for the two catchments, (ii) its ability to simulate the
10 patterns observed in groundwater (water level and chemistry) at the same temporal scales, and (iii)
11 the consistency between the nitrate transit times resulting from the model calibration and the
12 general pattern of the probable nitrate concentration evolution in the region since the beginning of
13 the intensification of agriculture in the 1960s. Finally, the sensitivity of the simulated transit times to
14 parameter uncertainty and climate variability was assessed and discussed by comparison with results
15 obtained using other methods such as physically based models or groundwater dating with
16 atmospheric tracers (CFCs).

17

18 **2. Materials and methods**

19

20 **2.1 Study catchments**

21 Kerbernez (11.5 ha) and Kerrien (10.5 ha) are two adjacent headwater catchments in South-western
22 French Brittany (47°, 35' N; 117°52' E, see Figure 1). They belong to the Observatory for Research on
23 Environment AgrHys (http://www6.inra.fr/ore_agrhys_eng/) which is part of the French network of
24 catchments RBV (<http://rnbv.ipgp.fr/>). Elevations range from 14 to 38 m.a.s.l., and slopes are less
25 than 8.5%. The climate is oceanic with mean annual temperature of 11.9°C, minimum of 5.9°C in

1 winter and maximum of 17.9°C in summer. Both catchments are underlain by the same bedrock, a
2 fissured and fractured granite, the “leucogranodiorite of Plomelin” (Bechennec and Hallegouet,
3 1999), overlain by a weathered regolith of 1 to more than 20 m depth. Soils are mainly sandy loam
4 (dystric cambisol as per FAO classification) with an upper horizon rich in organic matter (4.5-6%). Soil
5 depth proportions are given in Table 1. Soils are well drained except in the bottomlands which
6 represent 1% and 7% of the total area in the Kerbernez and Kerrien catchments respectively.

7 Agriculture dominates the land use with 40% of the total area in Kerbernez catchment and 86% in
8 Kerrien catchment. Urban area represents 27% of the Kerbernez catchment and 4% of the Kerrien
9 catchment. Dairy farming is the major agricultural activity. On average over the period 1992-2012, in
10 proportion of the total agricultural lands, grazed grasslands occupy 67% and 70% , maize 19% and
11 22%, and cereals 13% and 5% of the Kerrien and Kerbernez catchments respectively. Most of the
12 grasslands are intensively grazed, and maize and cereals receive substantial amounts of fertilizers
13 and manures.

14 **2.2 Environmental and agricultural monitoring**

15 Environmental and agricultural long-term time series are available at these sites. All the monitoring
16 points are indicated on Figure 1. Meteorological data (rainfall, net solar radiation, air and soil
17 temperatures, wind speed and direction) were recorded hourly since 1991 in an automatic weather
18 station (CIMEL) and daily Penman PET was calculated. Mean annual rainfall over the period 1992-
19 2012 is 1113 mm (+/-20%) and Mean annual Penman potential evapotranspiration (PET) is 700 mm
20 (+/- 4%).

21 Stream discharge was calculated from water level measurements at the outlets using V-notch weirs
22 equipped with a shaft encoder (OTT Thalimedes) recording every 10 min since 2000, and from weekly
23 data recorded manually from 1997 to 2000. Water was sampled at the outlet since 1991 from 2 to 4
24 times a month. Mean annual runoff is 272 mm (+/- 45%) and 360 mm (+/- 60%) at the Kerbernez and
25 Kerrien outlets respectively. In both catchments, base flow index is about 80 to 90%, thus the

1 hillslope aquifer is the main contributor to the stream for both flow and nitrates export (Molenat et
2 al., 2008; Ruiz et al., 2002a). The difference in runoff between the two catchments is attributed to
3 differences in hydraulic conductivity of the regolith (Martin *et al.*, 2006).

4 Piezometers were drilled in 2000 along two transects (Martin et al., 2004). For this study, simulation
5 results will be compared with the trends observed in upslope ($GW_{KB,Up}$ and $GW_{KN,Up}$) and midslope
6 ($GW_{KB,Mid}$ and $GW_{KN,Mid}$) piezometers for Kerbernez and Kerrien catchments respectively. All
7 piezometers consist of PVC tubes screened along the bottom meter of their length. The annular
8 space around the tube was filled with sand in the screened zone, with bentonite above the screened
9 zone and up to the last meter below ground surface, and then with concrete. This design aimed at
10 allowing the determination of groundwater chemistry at a relatively precise sampling depth.
11 Groundwater levels were monitored since 2001 either monthly if manual or every 15 min using
12 vented pressure probes (OTT, Orpheus Mini). Water samples were collected in the piezometers,
13 fortnightly during the recharge period and monthly otherwise, with an open bailer with a ball valve at
14 the bottom. Purging was carried out every 3 months.

15 All water samples (stream and groundwater) were filtered in the field with a GNWP 0.45 μ m filter
16 and stored in the dark at 4°C. Nitrate concentrations were analyzed in the laboratory by ionic
17 chromatography (Dionex).

18 An extensive survey of agricultural practices was carried out every year since 1992 to calculate the
19 amount of annual N available for leaching on every agricultural plot of the site (Ruiz et al., 2002a).
20 Collected data included the crop type, the amount and type of inorganic and organic fertilizers, the
21 crop yields, management of crop residues and the dates of all technical operations. For grasslands,
22 the proportion of legumes and the management of cattle grazing were documented.

23 **2.3 Flux computation**

24 2.3.1. Output fluxes computation

1 Daily stream flows were computed from the 10 min records. As this study is focused on base flow,
2 individual storm events were discarded and base flow was estimated by linear interpolation.
3 Instantaneous flow values were labeled as storm event when precipitations during the 24 previous
4 hours were greater than 5 mm. This threshold and the 24-hour period were deduced from the
5 analysis of the flood events (results not shown). Indeed, over the period 2001-2010, average flood
6 event duration was about 12.5 hours, and the threshold of cumulated precipitation to induce a storm
7 flow has been estimated around 5 mm. Annual water and NO₃-N fluxes were computed using a linear
8 interpolation of NO₃ concentrations between two sampling dates, *i.e.* over 1 to 4 weeks. According to
9 (Moatar and Meybeck, 2005), this method warrants an estimation of annual nitrate load with a
10 precision ranging between 10 % (15 days sampling frequency) and 15 % (monthly sampling
11 frequency) for their study site: the Loire River. Other comparison studies also showed that this
12 method leads to a good compromise between error and bias (e.g. Zamyadi et al., 2007).

13 2.3.2. Soil Nitrogen balances computation

14 For arable land, the annual amount of N available for leaching was computed using the budget model
15 proposed by COMIFER (1996) for France and parameters were modified according to regional
16 references for the fate of N in organic fertilizers (Chauvin et al., 1997) and according to local
17 references from lysimeter experiments carried out in the catchment to quantify the mineralization of
18 soil organic matter (Simon and Le Corre, 1992). Positive budgets were then aggregated in each
19 catchment for all the fields, to calculate the annual amount of N available for leaching at the
20 catchment scale. More details about the method can be found in Ruiz et al. (2002a), but we modified
21 the calculation for grazed grasslands and for mineralization following grassland destruction, by using
22 the more recent model Territ'eau (Gascuel-Oudoux et al., 2009) which is now used as a decision-
23 support tool by stakeholders in the region ([http://agro-transfert-bretagne.univ-
24 rennes1.fr/Territ_Eau/](http://agro-transfert-bretagne.univ-rennes1.fr/Territ_Eau/))

1 Drainage and nitrogen leaching through the soils were simulated for each catchment using the
 2 annual N available for leaching for each catchment, meteorological data series (rainfall and PET), and
 3 soil properties (depth and water holding capacity) as input data, and using the Burns' model of
 4 nitrate leaching (Burns, 1975) at a daily time step. We considered soil layers 10 cm deep. Water
 5 holding capacity was set to 20 mm in the 30 first cm under soil surface, and to 15 mm below. The
 6 Burns model was applied at the scale of the catchment, by using the proportion of soil depths in the
 7 catchment area (Table 1).

8 **2.4 Model rationale**

9 2.4.1. Hydrology

10 In lumped models, the worth of using two reservoirs for simulating slow and fast groundwater
 11 components has been illustrated in several studies (e.g. Clark et al., 2009; Stewart et al., 2010;
 12 Woodward et al., 2013). In this paper, we used a modified version of the model ETNA (Ruiz et al.,
 13 2002b), which simulates water flow and nitrate concentrations at the outlet in base flow conditions
 14 as the result of two parallel linear groundwater reservoirs characterized by different recession
 15 coefficients α (in day⁻¹). A conceptual representation of this simple model is proposed in Figure 2.
 16 Each reservoir is fed by the drainage D (mm) computed using the Burns model (see section 2.3). The
 17 mobile water volume in each reservoir V_i (mm) and corresponding outflow Q_i (mm) are expressed as
 18 follows:

$$19 \quad V_i(t) = V_i(t - dt) - Q_i(t - dt) + D(t) \quad (\text{Eq. 1})$$

$$20 \quad Q_i(t) = \alpha_i V_i(t) \quad (\text{Eq. 2})$$

21 where t is the time (day), and dt the time step equal to 1 day.

22 Both study catchments lie on a deep regolith layer which allows a deep groundwater flow below the
 23 catchment outlets, leading to a systematic deficit in the water annual balance. This is accounted for
 24 in the model by the parameter f_i (in %) representing the proportion of deep losses. A pond has been

1 settled in 2006 near the outlet of the Kerbernez catchment, and is partly draining the stream flow. It
 2 explains the difference of minimal base flow values after this date. Therefore the value of f_i was
 3 calculated from the difference between observed discharge and computed recharge (Burns model)
 4 on the period 2001-2011 and 2007-2011 for Kerrien and Kerbernez catchments respectively. It was
 5 57.35% and 54.55% for Kerrien and Kerbernez catchments respectively. For all the calibrations on
 6 stream flow values, we considered only this latest period of the data series for both catchments. The
 7 stream flow is the combination of the contributions of the two groundwater reservoirs which evolve
 8 independently. The relative part of each reservoir (f_{fast} and $1-f_{fast}$) is specific to each catchment.
 9 Considering the deep losses and the 2 reservoir contributions, the stream flow is finally computed as:

$$10 \quad Q(t) = (1 - f_i)(f_{fast}Q_{fast}(t) + (1 - f_{fast})Q_{slow}(t)) \quad (Eq. 3)$$

11 where Q is the stream flow, Q_{fast} and Q_{slow} the outflow from fast and slow stores respectively, all
 12 expressed in $\text{mm}\cdot\text{day}^{-1}$.

13 2.4.2. Nitrogen transfer

14 Nitrogen input to each reservoir is the daily nitrogen leaching L ($\text{kg N}\cdot\text{ha}^{-1}$) computed using the Burns
 15 model (see section 2.3.2). To take into account the microporosity of the regolith, a constant
 16 immobile volume of water was considered. We assumed a complete mixing between mobile and
 17 immobile water within a day, leading to the following equation for each store:

$$18 \quad C_i(t) = k \frac{S_{Ni}(t-dt) + L(t) - Q_i(t-dt)C_i(t-dt)}{V_i(t) + V_{im,i}} \quad (Eq. 4)$$

$$19 \quad S_{Ni}(t) = C_i(t)(V_i(t) + V_{im,i}) \quad (Eq. 5)$$

20 where for each reservoir i , S_{Ni} is the N storage, ($\text{kg N}\cdot\text{ha}^{-1}$), $V_{im,i}$ the immobile volume (mm) and C_i the
 21 concentration (in $\text{mg NO}_3\cdot\text{L}^{-1}$). k is a mass conversion factor (from $\text{kg N}\cdot\text{ha}^{-1}$ to $\text{mg NO}_3\cdot\text{L}^{-1}$, $k=443$).

22 The daily concentration at the outlet (C_{out} , in $\text{mg NO}_3\cdot\text{L}^{-1}$) is then computed according to the relative
 23 contribution of the 2 reservoirs:

$$1 \quad C_{out}(t) = \frac{(1-f_l)(f_{fast}C_{fast}(t)Q_{fast}(t) + (1-f_{fast})C_{slow}(t)Q_{slow}(t))}{Q(t)} \quad (Eq. 6)$$

2 The nitrate concentration of the deep losses is assumed to be equal to the concentration at the
 3 outlet. Denitrification has been largely investigated in various surface reducing environments. It has
 4 also been assumed in groundwater, once oxygen concentrations have decreased. However no
 5 transformation is assumed to occur in the groundwater or in the unsaturated zone, the main reason
 6 being the high dissolved oxygen content of groundwater in the region (Bidois, 1999). In the study
 7 site, the average concentration is about 5 mg of dissolved O₂/l (unpublished data). Indeed, a regional
 8 investigation of groundwater ages and nitrate concentrations (Aquilina et al., 2012) showed only
 9 limited evidences for denitrification. Evidences for denitrification were observed in the region only in
 10 specific environments such as pyrite-rich schists (Pauwels et al. 2000) or in sites influenced by
 11 pumping (Tarits et al., 2006). One explanation might be related to the passivation of reactive surfaces
 12 of reduced minerals inhibiting autotrophic denitrification processes. In the Kerrien and Kerbernez
 13 catchments, although potential denitrification was assumed in the unsaturated zone (Legout et al.,
 14 2005) no denitrification process has been evidence through chemical or isotopic analysis (Martin et
 15 al., 2004; Pierson-Wickmann et al., 2009).

16 2.4.3. Nitrogen transformation in the riparian-riverine zone

17 The biological transformation of nitrate, either by denitrification in the wetlands or stream
 18 consumption by aquatic primary producers was simulated as a constant daily amount of nitrate
 19 removal R_c (kg N.ha⁻¹.day⁻¹). Main limiting factors are known to be the nitrate availability, the
 20 temperature, the soil moisture, and light (Billen et al., 1994; Oehler et al., 2009). These factors vary
 21 seasonally and to some extent are likely to compensate each other, e.g. in winter, wetland saturation
 22 favors the occurrence of anoxic conditions and nitrogen concentration are often larger, while in
 23 summer temperature and light intensity favor biological activity. In addition, even if nitrate removal
 24 was higher in winter, the effect on nitrate concentration would be negligible considering the large

1 nitrate load. Thus, simulating biological removal as a constant appeared as a reasonable
2 parsimonious assumption. The stream nitrates concentration was finally computed as:

$$3 \quad C(t) = \frac{\text{Max}[kC_{out}(t)Q(t) - R_c; 0]}{Q(t)} \quad (\text{Eq. 7})$$

4 The model was run using a Matlab® code (R2008a version).

5

6 **2.5 Model calibration**

7 In order to optimize the parameter set values and determine the associated uncertainties, we
8 applied the Global Likelihood Uncertainty Estimation (GLUE) framework proposed by (Beven and
9 Binley, 1992). The GLUE method is an uncertainty analysis technique inspired from Bayesian
10 averaging of models and predictions, *i.e.* where a prior distribution of models is assessed in terms of
11 some likelihood measure relative to the observations, and a posterior distribution is then calculated
12 (Beven and Freer, 2001). A preliminary manual calibration was performed, and allowed us to define a
13 prior range of values for each parameter. The initial conditions were fixed at this stage and were not
14 integrated into the GLUE procedure, as a preliminary test showed that they have little impact on
15 GLUE results. Then 10,000 runs were performed using a random uniform sampling procedure (as no
16 prior knowledge was available on the parameter distributions). The independent calibration for each
17 catchment resulted in large uncertainties of the 6 calibration parameters. To reduce this uncertainty
18 we calibrated the model simultaneously on the two catchments, using combined likelihood and
19 assuming that only two parameters can vary between the two catchment: the proportion of fast
20 store (f_{fast}) and the nitrate removal constant (R_c). The recession coefficients in each store (α_f and α_s)
21 and the immobile volumes in each store ($V_{im,f}$ and $V_{im,s}$), which are intrinsic characteristics of the
22 regolith aquifer, were assumed to have the same value in both catchments. This assumption was
23 supported by previous model calibrations for these sites (Ruiz et al., 2002b) and seems reasonable in
24 the case of neighboring catchments in the same geological context. Calibration was carried out in

1 two steps, adjusting first the hydrological parameters (α_f , α_s , $f_{fast,KRN}$, $f_{fast,KBZ}$) only, and then the
 2 parameters related to the N transformations and transfers ($V_{im,f}$, $V_{im,s}$, $R_{C,KRN}$, $R_{C,KBZ}$).

3 Several performance criteria based on the distance between simulated and observed values were
 4 tested to build the objective functions (OF). The mean absolute error (MAE), which is a classical
 5 standard in model performance assessment (Bennett et al., 2013) was finally selected.

6 We chose to apply this criterion on the square root rather than on the absolute value of stream flow
 7 as our simplified model does not simulate the flood events but only the base flow dynamics. For a
 8 detailed discussion on this point, one can see in Pushpalatha et al. (2012) a study of the relevance of
 9 a range of classical criteria in hydrology to assess the performance of models in reproducing low flow
 10 or medium flow. For the nitrogen related components of the model, the same criterion, applied on
 11 the square root of nitrate concentrations, was also found to be the most relevant.

12 The MAE is always greater than 0 and tends to infinity as the discrepancy between model and
 13 observations increases. An objective function (OF) value for each catchment j and each parameter set
 14 θ was obtained by taking the inverse of the normalized MAE so that the OF increases with the model
 15 performance:

$$16 \quad OF^j(\theta) = \left(\frac{MAE^j(\theta)}{\sum_{\theta} MAE^j(\theta)} \right)^{-1} \quad (Eq. 8)$$

17 For each calibration step, the combined likelihoods, $CL(\theta)$, were computed as the product of the
 18 objective functions obtained for both catchments divided by their sum for all explored parameter
 19 sets:

$$20 \quad OF(\theta) = \prod_{j=1}^{nb \text{ of catchments}} OF^j(\theta) \quad (Eq. 9)$$

$$21 \quad CL(\theta) = \frac{OF(\theta)}{\sum_{\theta} OF(\theta)} \quad (Eq. 10)$$

22 Thus, the combined likelihoods are between 0 and 1 and increase with the model performance.

1 The 0.1% of the running sets giving the best CL was selected as behavioral parameter sets. The
 2 optimal value of each parameter was taken from the average of the behavioral set values, and the
 3 uncertainty was computed as the relative standard deviation of the behavioral set values.

4

5 **2.6 Evaluation of model consistency on groundwater observations and long time series**

6 We evaluated the model consistency on its ability at simulating the groundwater dynamics, and then
 7 its compatibility with realistic long-term evolution trends. To compare simulated groundwater
 8 storage to observed water table levels in piezometers, simulated water table levels were calculated
 9 from the simulated content of water in the two reservoirs and by using a value of 1.5% for the
 10 specific yield of the regolith, that was found on this site by Martin et al., (2006), and a maximum
 11 mobile volume of groundwater of 650 mm. The average daily simulated water table levels were
 12 calculated as follow:

$$13 \quad Z_{sim}(t) = -1/0.015 \left(650 - \left(f_{fast} V_{fast}(t) + (1 - f_{fast}) V_{slow}(t) \right) \right) \quad (Eq.11)$$

14 In order to test if such a simple lumped model, calibrated only on recent years, could simulate
 15 realistic dynamics over a longer period, the model was applied on both catchments for the period
 16 1965-2011, corresponding to the full available meteorological time series on the site, using the mean
 17 behavioral parameter set. We considered that starting the simulation with nitrate-free reservoirs in
 18 1965 was realistic as all available regional data on stream nitrate concentration before 1975 display
 19 very small values, all smaller than 10 mg NO₃/L (Aquilina et al., 2012). As mentioned earlier, no
 20 quantitative information on the agricultural practices in the catchment is available before 1992.
 21 Regional or national statistics are difficult to downscale for our headwater catchments, which
 22 comprise only few plots. Therefore, we generated time series of annual N available for leaching using
 23 very simple assumptions relying on historical regional statistics (Aquilina et al., 2012) and qualitative
 24 information on the studied catchments.

1 Although no quantitative agricultural data is available before 1992, interviews with farmers revealed
2 that while in the Kerrien catchment, farming has been consistently primarily dedicated to dairy
3 production at least since the 50's, the Kerbernez catchment used to host a farm with very intensive
4 pig production. The main agricultural field in this farm was devoted to outdoor sows. Farm buildings
5 included indoor pigs for fattening, pig slurry being spread mostly in the catchment itself before 1990.
6 The pig production decreased progressively. Outdoor sows stopped in the 80's and indoor pigs
7 declined slowly in the 90's until 2000 when the farm ceased its activity and its agricultural fields were
8 allocated to neighboring dairy farms.

9 The simplest evolution which can be generated is a linear increase of annual N available for leaching
10 starting at zero in 1965, and increasing until 1992, as a result of the increase of N fertilizers and
11 animal production. This simple scenario was applied on the Kerrien catchment as we know that it
12 was consistently dominated by dairy farming throughout the period. The slope of the increase was
13 adjusted to fit the observed concentration at the outlet at the beginning of the observation period in
14 1992. Figure 3 presents the full time series of annual N available for leaching, with estimated values
15 from 1965 to 1991 and observed values after 1992.

16 For the Kerbernez catchment, the observed seasonal dynamics (inverse correlation between stream
17 flows and nitrate concentrations) imposed that the slow store must be more concentrated in nitrate
18 than the fast store in the latest period. Considering the model structure, this was possible only if the
19 annual N available for leaching had increased dramatically in the initial period, and then started
20 decreasing before 1992. This was consistent with the qualitative information collected in this
21 catchment that revealed the early development of pig intensive breeding and its progressive decline
22 since the 1980s. The two slopes of these successive trends of annual N available for leaching, starting
23 at zero in 1965, were therefore adjusted to reach the observed concentration in 1992. The full time
24 series (Figure 3) showed that the adjustment led to very high values of annual N available for
25 leaching in the 60s. These high values might not been realistic, and a smoother, more plausible

1 scenario might have been designed, for example by introducing three successive trends (increase,
2 stabilization, decrease). However, as no data were available to constrain such a scenario, and as the
3 simulated concentrations in the latest period were little sensitive to the distribution of annual N
4 available for leaching in the early period, this option was discarded.

5 **2.7 Nitrate transit time calculations**

6 Transit time distribution of water or solutes is the result of the highly variable flow pathways
7 followed by individual molecules in catchment. While being often considered as an intrinsic property
8 of a catchment, transit times are also depending on the climatic conditions.

9 The calibrated models were used to estimate the transit time of nitrogen in each catchment for the
10 observed climate time series 1965-2012, by simulating the time needed for a pulse of nitrate (a single
11 day pulse input of nitrate in a nitrate-free system) to exit the catchment. For this exercise, the
12 parameter for nitrates removal (R_c) was set at zero, to ensure a conservative behavior of nitrate, and
13 nitrate output was calculated as the sum of stream and deep loss outputs. In order to estimate the
14 sensitivity of transit time estimations to climatic conditions, three dates for the nitrate input pulses
15 were selected in the observed climatic time series, on 1st of August 1968, 1974 and 1980,
16 representative of dry, average and wet climatic sequences respectively. For each of these cases, the
17 model was run for each of the behavioral sets of parameters so that it was possible to estimate the
18 sensitivity of the nitrogen transit time to parameter uncertainties. It has to be noted that as the
19 estimated transit time assumes conservative transport of nitrate and consider all nitrate leaving the
20 system, and may differ from actual transit time of nitrate leaving the system specifically via the
21 streams.

22 The transit time distribution can be represented as the evolution with time of the cumulated fraction
23 of recovered tracer in output flux. The Mean Transit Time (MTT) is the most commonly used proxy
24 for this distribution but as a full tracer recovery is needed to calculate it, it frequently imposes a
25 repetition of available climate time series, e.g. (Dunn et al., 2007). However water transit time and

1 nitrate transit times may differ slightly depending on the flow pathways taken by the nitrate
2 molecules, e.g. in the study catchments storm events are a fast flow pathway responsible for the
3 lowest values in the water transit time distribution curves while they are nitrate free as storm runoff
4 occurs mainly over saturated area and so when nitrate has already been infiltrated (e.g. Aubert et al.,
5 2013). As we wanted to characterize the effect of real climate on nitrate transit time estimations, we
6 rather used Half Nitrogen Recovery Time (HNRT), which correspond to the times required to recover
7 half of the nitrate introduced with the input pulse. Values of HNRT are slightly smaller than MTT
8 when the distribution presents “long tails”, but they still can be used to compare with groundwater
9 MTT found in the literature.

10

11 **3 Results and discussion**

12 **3.1 Model calibration**

13 The simultaneous calibration of the model on the two paired watersheds using the GLUE
14 methodology, on 8 parameters using 4 observed times series (stream discharge and nitrate
15 concentration for the two catchments) led to a reasonably low uncertainty on parameters estimation
16 (Table 2, Figure 4). The parameters describing the aquifer properties (α_f , α_s , $V_{im,f}$, $V_{im,s}$) appeared well
17 constrained with an uncertainty of about 10%. The removal constants R_o are more uncertain
18 especially in the Kerrien catchment, but their values being very low, they have little impact on the
19 stream concentration except during very low flow periods. The proportion of fast store f_{fast} in each
20 catchment, which controls the contrast between the catchment responses, reflects the more reactive
21 nature of Kerrien compared to Kerbernez. For Kerrien, the behavioral value of f_{fast} tends to the upper
22 bound which has been fixed to impose a maximum of 90% of each store, and its uncertainty is also
23 relatively large.

1 Model performance was assessed with classical statistical criteria (Dawson et al., 2007): the Nash
2 Sutcliffe Efficiency criteria on square root discharge were 0.73 and 0.81, and the root mean square
3 error on concentration were 8.5 and 13.9 mg.l⁻¹ for Kerrien and Kerbernez catchments respectively.
4 The simulated ranges of base flow discharge and nitrate concentration corresponding to behavioral
5 parameter sets were relatively narrow (Figure 5). For discharge, they fit better the medium and low
6 flow values than the high flow values, which is linked to the properties of the criterion used in the
7 objective function. For nitrate concentration, simulated seasonal variations obtained with behavioral
8 sets were in good agreement with the observed ones in the two catchment (Figure 5), both in terms
9 of dynamics (cycles with flow-concentration relationship that are negative in Kerbernez and positive
10 in Kerrien) (Martin et al., 2006) and in terms of amplitude, Kerbernez being more buffered than
11 Kerrien catchment. In the model, these seasonal patterns in nitrate concentrations were mostly
12 reflecting the nitrogen status of the catchments, through the relative nitrate concentrations in fast
13 and slow stores indicating whether in the long term the groundwater N storage was increasing
14 (Kerrien) or decreasing (Kerbernez). Nitrate removal was not affecting noticeably the concentration
15 dynamics except during very low flows periods in Kerrien catchment, and had very little effect on
16 nitrate load, the removal of nitrogen representing less than 1% of the total nitrogen output flux on
17 the whole period. The observed interannual trends were also reasonably well reproduced by the
18 simulations in both catchments. In Kerrien, concentrations increased slightly during the first years,
19 declined markedly from 2000 to 2004, increased again up to 2007 and finally declined slightly. In
20 Kerbernez, the concentration declined markedly along the whole period, except for a period of
21 relative stabilization between 2003 and 2008.

22 However, when looking closely at yearly fit between observed and simulated concentrations, some
23 discrepancies appeared which can be due to the model itself and also to errors in measurements of
24 outputs and on forcing variables. Uncertainties on input data are crucial, especially when these data
25 are coming from agricultural practices estimation. (Howden et al., 2011a) show that their model
26 ability to simulate long-term nutrients concentration depends principally on the precision on the

1 sources. In our work, we did not study specifically the uncertainty on the input data and its
2 propagation on the model outputs. The comprehensive agricultural survey led to estimations of
3 annual N available for leaching that were consistent with lysimetry studies performed in the same
4 site (Simon and Le Corre, 1992). However, as the Burns model simulates highly variable inputs of
5 nitrogen leaching during the whole recharge period, this could impose large values of immobile
6 volume in groundwater stores for buffering this variability. To test this hypothesis, an additional
7 calibration with GLUE was tested, using the same annual N available for leaching but constant annual
8 concentrations of nitrates in the drainage water. Optimal parameter values and associated
9 uncertainties were finally not significantly different (results not shown), indicating that the model
10 was more sensitive to the average of N input over several years than to short term variations.

11 It is worth noting that discrepancies were often linked with inaccurate simulations of discharge (e.g
12 in the years 2005 and 2006 in the Kerrien catchment, where the largely overestimated nitrate
13 concentrations corresponded to largely overestimated stream discharges). Calibration on each
14 catchment independently would have provided a slightly better fit, but at the cost of a much larger
15 parameter uncertainty (results not shown). This highlights the difficulty of model calibration
16 procedures to deal with seasonality. Most calibration effort in hydrology has been made on
17 reproducing peak flows, often event per event, and we miss relevant metrics to include seasonal as
18 well as inter-annual trends in performance criteria. The difficulty to find behavioral
19 parameterizations fitting pluriannual and seasonal trends simultaneously has been raised by (Choi
20 and Beven, 2007). When dealing with water quality, it becomes even more complex as its seasonality
21 is related to the combined dynamics of stream flow and solute transport. For example, in periods
22 when discharge is not well simulated, we should expect that a behavioral model would badly
23 simulate the concentrations, while calibration procedures will favor models that tend to compensate
24 the error on discharge by minimizing the concentration seasonality. This study showed that
25 calibrating the model simultaneously on two catchments displaying contrasted seasonal patterns in
26 concentration was one way of addressing this problem. The fact that it resulted in only one

1 parameter (f_{fast}) controlling the contrast between the catchment responses is also interesting, as this
2 parameter could serve as a useful metric to compare catchments in homogeneous geological
3 contexts.

4 **3.2 Assessment of model consistency**

5
6 The model consistency was assessed first by comparing simulated groundwater storage and nitrate
7 concentration with observed water level and nitrate concentration in piezometers then by testing its
8 capacity to represent realistically long term trends in nitrate concentration.

9 The observed nitrate concentration displayed contrasted dynamics according to the piezometer
10 location, as shown in Figure 6 only for two piezometers in each catchment for the sake of clarity. This
11 spatial variability, both vertical and lateral has been highlighted in (Legout et al., 2007; Rouxel et al.,
12 2011) in the same site. Simulations with the lumped model were not expected to fit individual
13 piezometers. This is one of the limits of using the same parameterization for the regolith properties,
14 here the specific yield, for the two catchments. However, comparing daily water table levels
15 observed in the upslope piezometers and simulated with the mean behavioral parameter set
16 provided interesting insights about the modeled behavior in the groundwater. Figure 6 showed that
17 groundwater dynamics were better reproduced for Kerbernez than for Kerrien. Simulated
18 concentrations were also showing contrasted dynamics in the different stores, the fast stores
19 displaying more marked seasonality and larger year to year variations. Interestingly, concentrations
20 dynamics observed in the hillslope appeared in good agreement with simulations of the fast store in
21 Kerrien catchment, and with simulations of the slow store in Kerbernez catchment. Therefore, these
22 results confirmed the consistency of the conceptual representation of the system, which led to
23 calibrated groundwater system largely dominated by the fast store in Kerrien and by the slow store
24 in Kerbernez.

1 Considering the very large immobile water volumes in each store resulting from the calibration
2 (Table 2), it was crucial to test if the resulting very large memory effect was compatible with long-
3 term trends of nitrate increase since the beginning of agriculture intensification. The long-term
4 simulations on the two catchments (Figure 7) based on the reconstructed times series of annual N
5 available for leaching (Figure 3) and using the mean behavioral parameter set, confirmed that the
6 calibrated models were reactive enough to reach the range of concentrations observed in 1992 even
7 when starting with a nitrogen free system in 1965. The concentrations in both groundwater stores
8 obtained in 1992 with these scenarios were close to the initial values that were imposed in the
9 calibration process for the two catchments (Table 3), leading to simulations of the years following
10 1992 that were very close to the ones obtained during calibration, in particular on Kerrien
11 catchment.

12 Figure 8 also illustrates the asynchronous dynamics of the simulated nitrate concentration in the fast
13 and slow groundwater stores, which are responsible for the seasonal patterns of nitrate
14 concentration in the stream. In the Kerbernez catchment, the inversion of concentrations between
15 fast and slow stores in the late 1980s, induced by the specific agricultural history of this site, led to
16 the inversion of the seasonal patterns of nitrate concentration in the stream in the later period.

17 These results showed that a simple conceptual lumped model, while remaining parsimonious and
18 allowing for a low uncertainty in parameter estimation, was not only able to simulate the observed
19 patterns of nitrogen variations at various time scales, from seasonal to pluriannual, but also proved
20 its consistency with respect to groundwater signature and long term behavior. This confirmed the
21 relevance of using parallel linear stores to represent groundwater in lumped conceptual models, e.g.
22 (Clark et al., 2009; Stewart et al., 2010; Woodward et al., 2013).

23 **3.4 Nitrate groundwater transit time**

24 The evolution with time of the cumulated fraction of tracer output flux for the range of the
25 behavioral parameters (Figure 8) shows that for the explored time series, the full tracer recovery was

1 attained after about 30 years for the Kerrien catchment while it was not attained even after more
2 than 40 years for the Kerbernez catchment, confirming that calculations of mean transit time was not
3 possible in this catchment using the available weather time series. The Table 4 shows that the
4 average Half Nitrogen Recovery Time (HNRT) was smaller in the Kerrien catchment than in the
5 Kerbernez catchment, while their variability was in the same range. Considering each pulse input
6 date separately reduced the standard deviation of HNRT in Kerrien and not in Kerbernez, showing
7 that in the Kerrien catchment, HNRT were more sensitive to the date of the pulse input than to
8 parameter uncertainty, while for the Kerbernez catchment both factors had a similar influence. In
9 terms of policy, it means that the time required to recover a good water quality will on such
10 catchments is going to be very long and strongly dependent on the meteorological conditions. Dry
11 years will tend to increase the time of nitrate storage in the catchment and then the recovery of low
12 concentration levels in streams. Moreover, in catchments characterized by longer nitrate transit
13 times (as the Kerbernez catchment) the predictions of N recovery under mitigation scenarios will be
14 more uncertain.

15 These results are consistent with a previous study in the same catchments which estimate from the
16 physically based model MODFLOW-MT3D that the reaction times required for equilibrating shallow
17 and deep parts of the aquifer were between 6 and 8 years for Kerrien catchment, and between 10
18 and 12 for Kerbernez catchment (Martin et al., 2006). (Molenat and Gascuel-Oudou, 2002) estimate
19 that transit times in a similar site (Kervidy-Naizin, also part of the ORE AgrHys) is ranging between 0
20 and 3 years, while arguing that it was still underestimated due to the fact that molecular diffusion
21 between macro and micro porosity was neglected, and because their estimation was based on
22 hydraulic gradients during winter high flows. A regional study using groundwater dating based on
23 CFC tracers, including some samples belonging to Kerrien and Kerbernez catchments, also found
24 water ages within the weathered aquifer that ranged between almost 0 to 20-25 years, with a mean
25 of 18 years (Ayraud et al., 2008). Although these studies are referring to different “times definitions”
26 like “reaction catchment times”, “catchment mean transit times”, or “groundwater age”, it is worth

1 noting that all these estimates are spanning the same range of values, which is larger than one year
2 and closer to the decade.

3 **3.5 Limitations and perspectives for generalization**

4 This work was based on very densely-monitored catchments, allowing to compare our model results
5 to various environmental variables (surface and groundwater), on long-term time series, and to
6 calibrate the model for two contrasted catchments simultaneously. The applicability of the approach
7 for catchments where monitoring is scarcer, particularly regarding surveys of agricultural practices, is
8 still to be tested. However, as we have shown that when nitrate transit time was large, the model
9 was not very sensitive to short term variations of nitrate leaching, rough estimations of agricultural
10 inputs might be enough to estimate catchment transit times with our approach.

11 The approach presented here is not restricted to the specific conceptual model that used in this
12 study, which is relevant for the case of headwater catchments where the hillslope groundwater is the
13 main contributor to the stream flow (Ruiz et al., 2002b). From the hydrological point of view, to deal
14 with other geological or climatic contexts or poorly gauged catchments, some adjustments may be
15 required. To achieve this, promising tools have been developed over the PUB decade (Hrachowitz et
16 al., 2013). To cite just a few examples of them (Gharari et al., 2011) proposed some approach for
17 model design based on catchment physical properties. (Fenicia et al., 2011; Kavetski and Fenicia,
18 2011) develop a flexible modeling approach to design improved conceptual models in adequation
19 with the data that are available to constraint it, and with multi-objective performance criteria (Euser
20 et al., 2013). Hrachowitz et al. (2014) highlighted how an iterative procedure of increasing process
21 complexity associated with the systematic use of catchment signatures, expert knowledge, and
22 realism constraints may guide the stepwise improvement of conceptual rainfall-runoff model to
23 increase its consistency.

24 Application on larger scales may also require some refined formulation of N transformation model.
25 (Montreuil et al., 2010) estimated that for order-6 streams in a similar context (Scorff basin, West

1 Brittany, France), up to 53 % of the annual nitrate flux during base flow can be removed within the
2 wetland or in the river. (Flipo et al., 2007) estimated, by using mean annual N balance over a
3 1200 km² catchment, that 20% of the nitrate flux leaving the root zone was removed by
4 denitrification. In such contexts, one can assume that riparian or in-stream transformations would
5 induce a superimposed seasonal cycle on nitrate concentration which would require more process-
6 based formalisms to discriminate the hydrological effect from the effect of others environmental
7 factors. For example, denitrification may involve some controls by the water saturation dynamics,
8 temperature, and the availability of substrates in wetlands (Pinay et al., 2007), while in-stream
9 processes will depend on the amount of nitrates, light, temperature conditions, and other nutrient
10 conditions (Garnier et al., 1995).

11 Nevertheless, the aim of this study was to assess the ability of such lumped models to provide
12 valuable estimates of transit times of agricultural-derived pollutants as nitrates within headwater
13 catchments in spite of parameter uncertainty and according to climate variability, and it proved their
14 interest when considering simultaneously contrasted catchments. In such an approach, the degree of
15 model complexity has to be consistent with the degree of contrast between the catchments, i.e. the
16 number of relevant processes which have to be represented.

17

18 **4 Conclusion**

19

20 In the present study, we showed that time series of nitrate concentration in streams can be used to
21 estimate nitrate transit times of more than several years in headwater catchments, by using a
22 lumped and parsimonious model. The performance and consistency of the model ETNA was found
23 adequate considering its ability to reproduce the nitrate concentration patterns (i) at different

1 temporal scales (seasonal and inter annual), (ii) both in the stream and the groundwater and (iii) for
2 the expected trends over several decades.

3 The GLUE approach was used to assess the parameter uncertainties and the subsequent
4 uncertainties on nitrogen transit times by calibrating ETNA on two paired research catchments where
5 contrasted chemical signatures are observed on a 20-year-period. Reasonably low parameter
6 uncertainties were obtained by calibrating simultaneously the two catchments. The effect of
7 parameter uncertainty on transit time estimations was in the same range as the effect of the climatic
8 variability. The estimated nitrate transit times were found consistent with results from other studies
9 in the same site or in the same region. Finally, only one parameter controlled the contrast in nitrogen
10 transit times between the 2 catchments, which could constitute a simple metric for comparing
11 catchments behavior in the same geological context.

12

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20

1 Table 1: Soil depth proportions in the two catchments

2

Proportions of soils depth in the catchment		
Soil Depth	Kerrien	Kerbernez
Less than 40 cm	33%	17%
40-50 cm	22%	60%
50-60 cm	10%	2%
60-70 cm	33%	11%
80-90 cm	2%	10%

3

4 Table 2: Mean values and uncertainties of behavioral parameters

5

Parameter	runs 10000		Objective Function				Unit
	Inferior Bound	Superior Bound	MAE(Q ^{1/2})		MAE(C ^{1/2})		
			Mean optimal parameter value	Uncertainty (%)	Mean optimal parameter value	Uncertainty (%)	
	f_{fast} KRN	0.1	0.9	0.865	2.8		
f_{fast} KBZ	0.1	0.9	0.229	36.84			1
α_{fast}	0.01	0.1	0.0252	11.22			d ⁻¹
α_{slow}	0.001	0.01	0.0079	13.42			d ⁻¹
$V_{im,fast}$	100	4000			2354	11.01	mm
$V_{im,slow}$	100	20000			16032	7.22	mm
R_c KRN	0	6			0.1728	70.21	10 ⁻² kg N.ha ⁻¹ .d ⁻¹
R_c KBZ	0	6			3.0766	16	10 ⁻² kg N.ha ⁻¹ .d ⁻¹

6

1 Table 3: Comparison of groundwater store concentrations in 1992 imposed for the model calibration
 2 and with the 1965-2012 simulation using the mean behavioural parameter set.

3

Concentration in GW store in 1992	Fast store		Slow store	
	calibration	Simulation 1965-2012	Calibration	Simulation 1965-2012
Kerrien	50	59.24	25	29.62
Kerbernez	95	61.17	120	125.8

4

5 Table 4: Half Nitrogen Recovery times and associated standard deviations among the range of
 6 behavioral sets and for each of the three input pulse dates, and for all the 30 simulations

7

Pulse	Volume (mm) of rainfall from tpulse to tpulse+8 years	Kerrien		Kerbernez	
		HNRT (y)	cv %	HNRT (y)	cv %
1	7813	4.8	11	12	15
2	8684	3.8	9	10	17
3	9225	3.4	8	10	22
All simulations		4.0	18.5	10.7	19.8

8

9

1

2 Figure 1: Localization of the study site and the monitoring points

3

4 Figure 2: Scheme of the modeling procedure using Burns and ETNA models

5

6 Figure 3: Time series of annual N available for leaching estimated from linear trend adjustment (1965-1991) and from yearly

7 farm surveys (1992-2012) in Kerrien (a) and Kerbernez (b) catchments.

8

9 Figure 4: Dotty plots of combined likelihood values and behavioral (CLOpt) sets depending on (a) hydrological parameters

10 and (b) on parameters related to nitrogen transfer and transformation.

11

12 Figure 5: Simulated and observed discharge (a, b) and nitrate concentration (c, d, in mg of NO_3/l) at the outlets of Kerrien

13 (a, c) and Kerbernez (b, d). BS are the results corresponding to behavioral parameter sets and mean BS are the results

14 obtained from mean behavioral parameter set values.

15

16 Figure 6: Simulated & observed water table depth (a, b: Z_{sim} , $GW_{KN,Up}$ and $GW_{KB,Up}$ are the groundwater depths

17 respectively simulated, measured on Kerrien and Kerbernez upslope piezometers) and nitrates concentration (c, d: C_{FAST} ,

18 C_{SLOW} , $GW_{KN,Up}$ and $GW_{KN,Mid}$, $GW_{KB,Up}$ and $GW_{KB,Mid}$ are the groundwater concentrations respectively simulated in fast and

19 slow stores, and, measured on upslope and midslope piezometers on Kerrien and Kerbernez) in groundwater of Kerrien (1a,

20 c) and Kerbernez (b, d) catchments. Simulation results correspond to the mean behavioral parameter set.

21

22 Figure 7: Long term trends simulation from 1965 to 2012: simulated and measured stream flow (a, b) & stream NO_3

23 concentration (c, d), and NO_3 concentration in groundwater reservoirs (e, f) on Kerrien (a, c, e) and Kerbernez (b, d, f)

24 catchments.

25

26 Figure 8: Evolution with time of the cumulated fraction of tracer output flux for the behavioural parameter sets for the

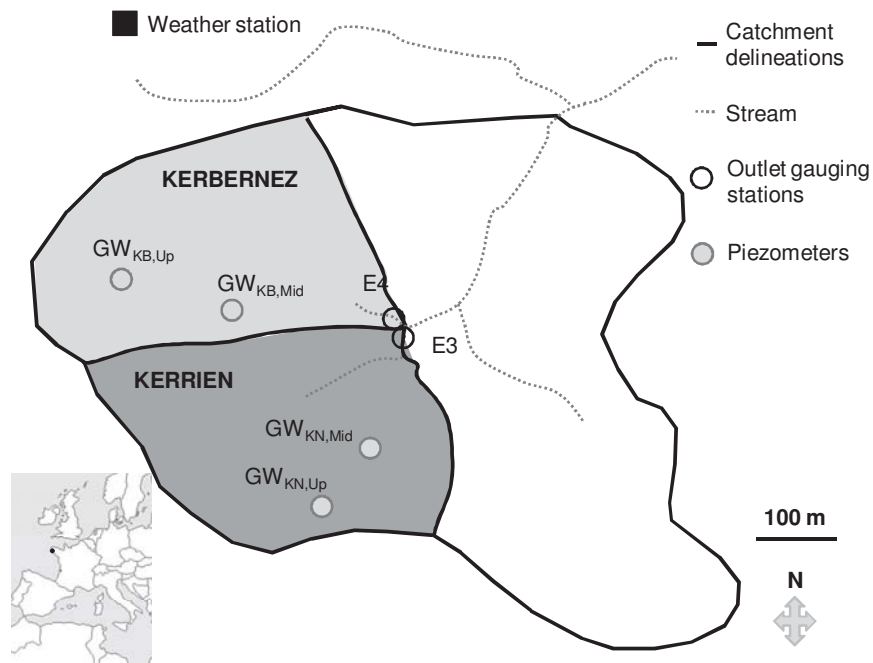
27 three different dates of pulse input application (pulses 1 to 3) in Kerrien (a) and Kerbernez (b) catchments. Cumulated

28 rainfall in the 8 years following the input date increase from pulse 1 to pulse 3 (see Table 4). Vertical dashed lines show the

29 half nitrogen recovery time (HNRT) for each case.

30

31

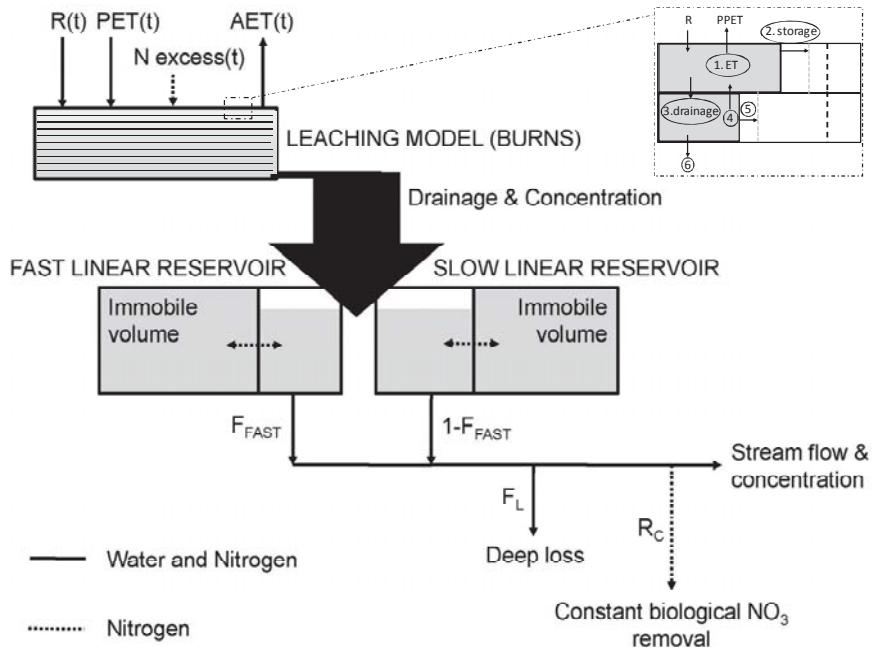


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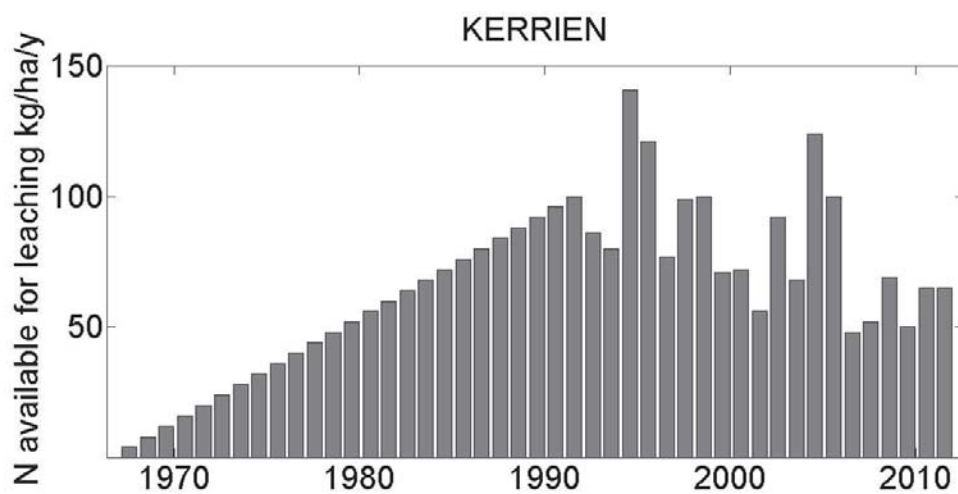
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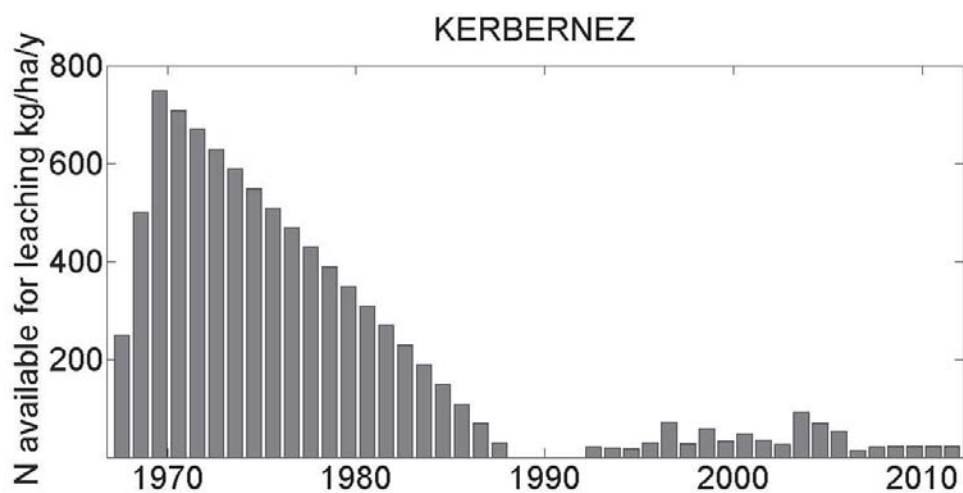
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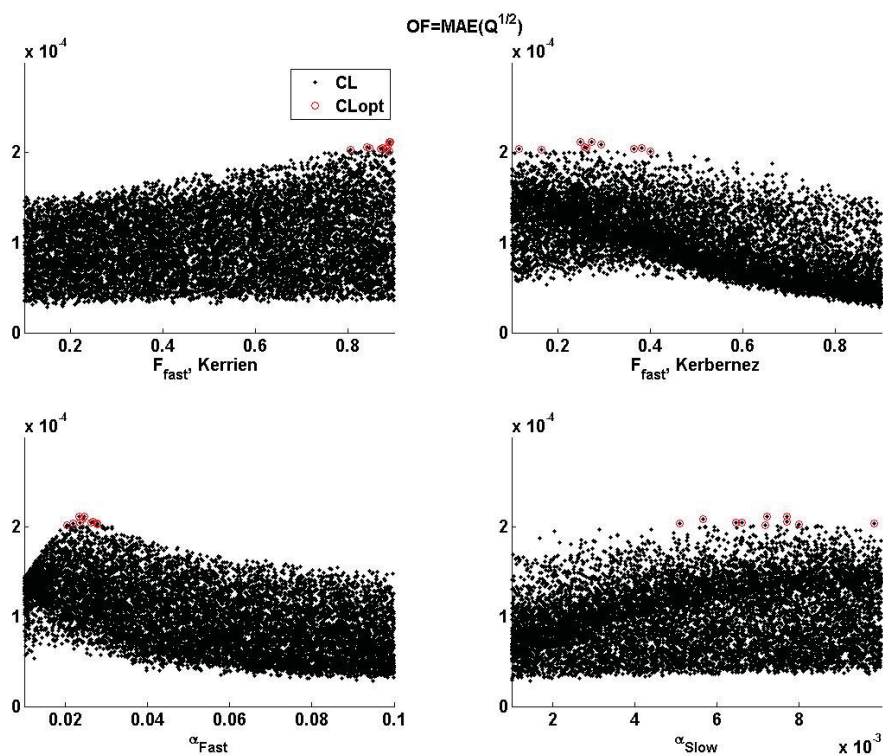
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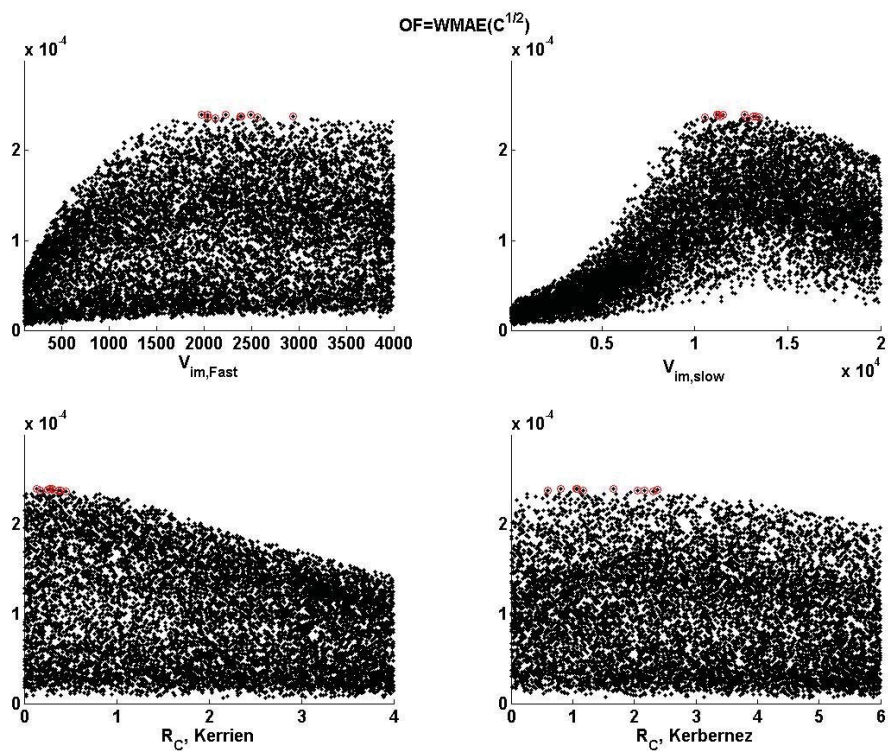
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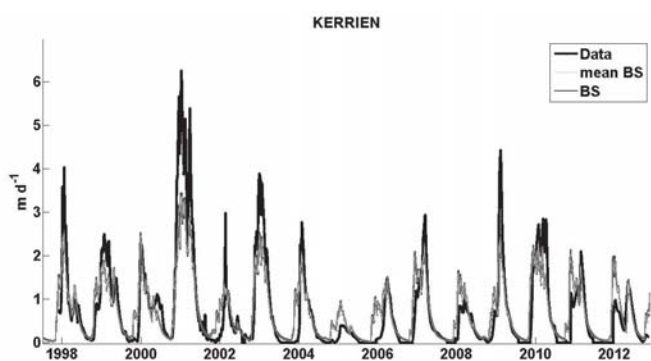


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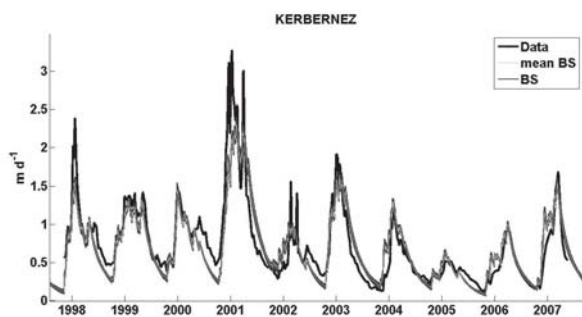


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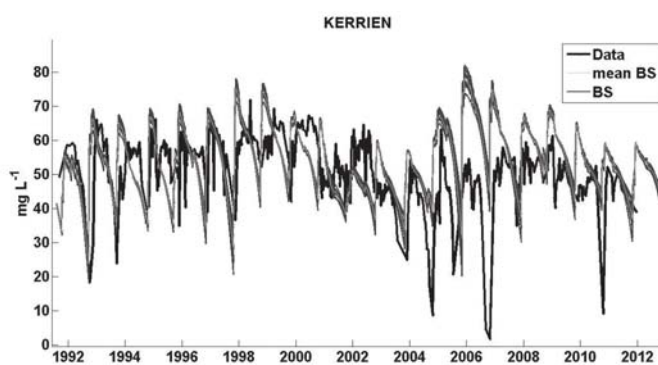
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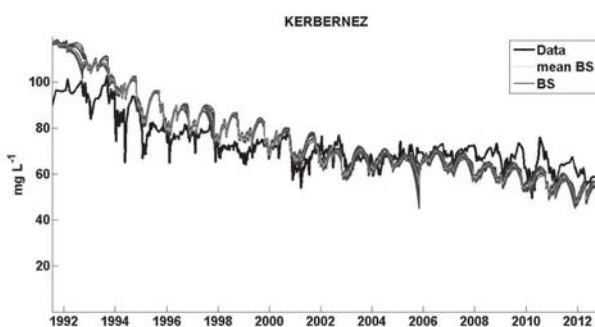
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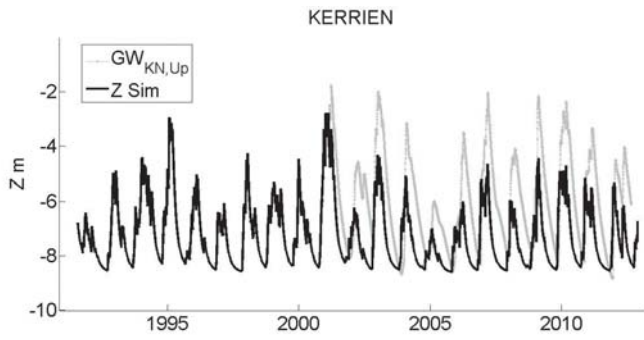
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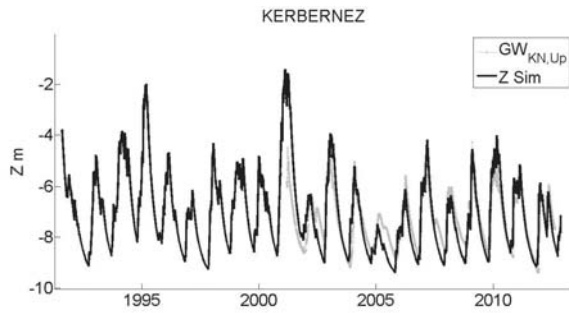
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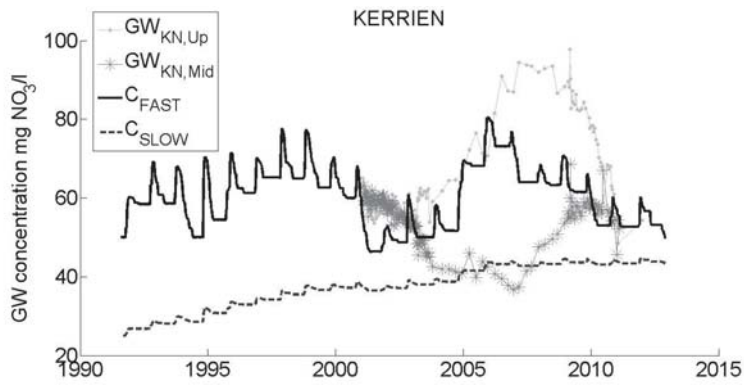
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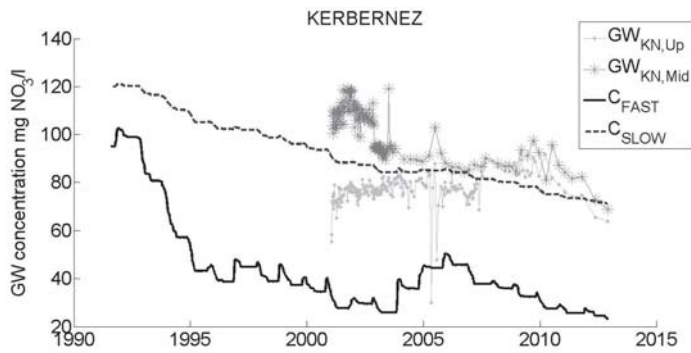
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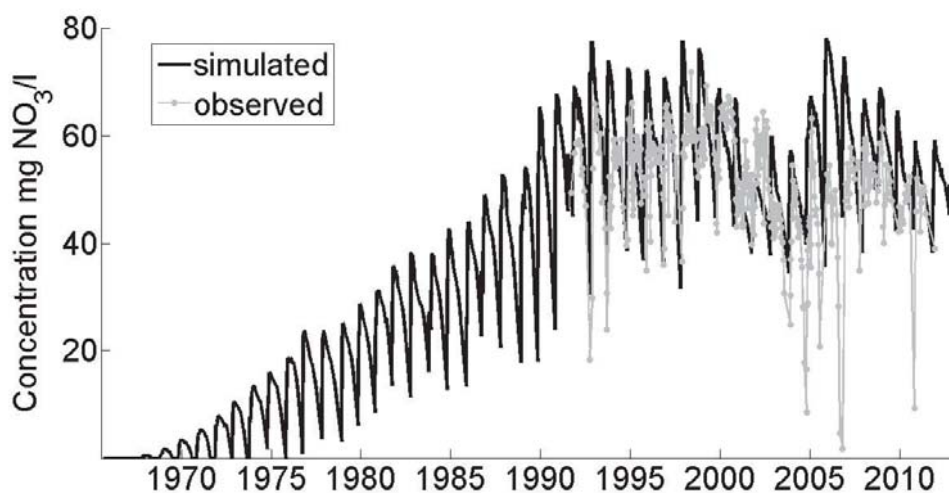
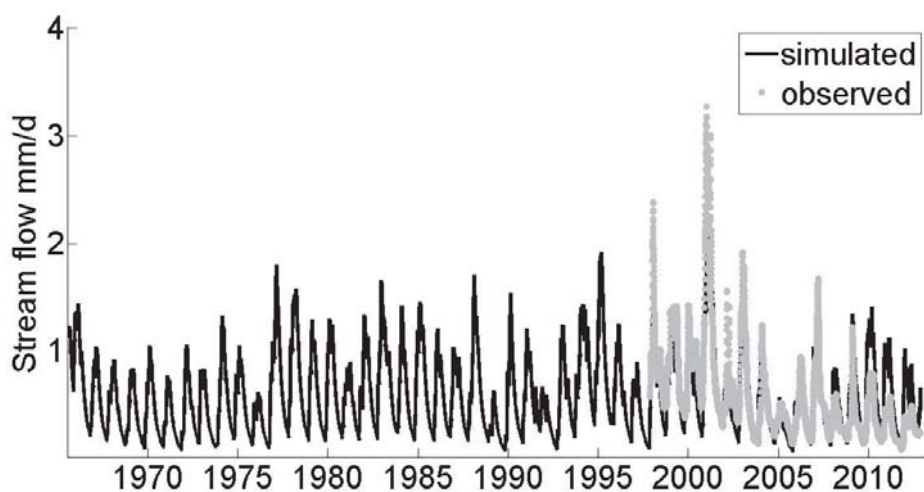
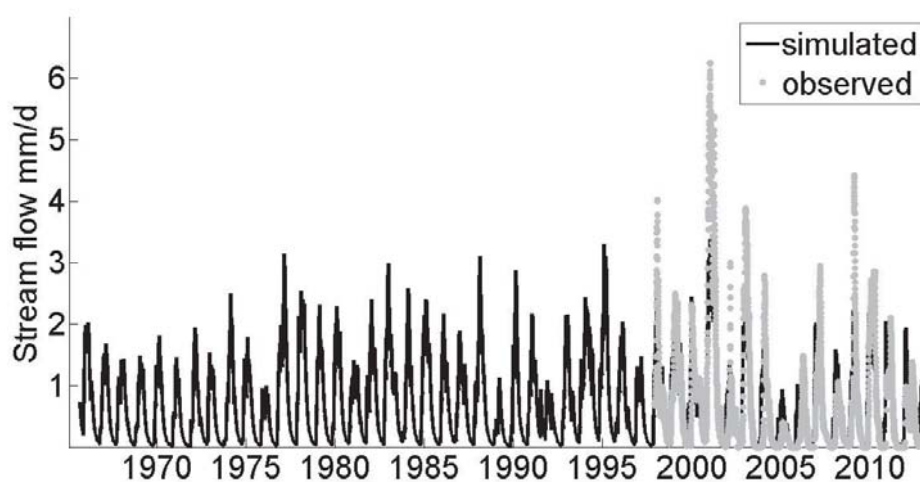
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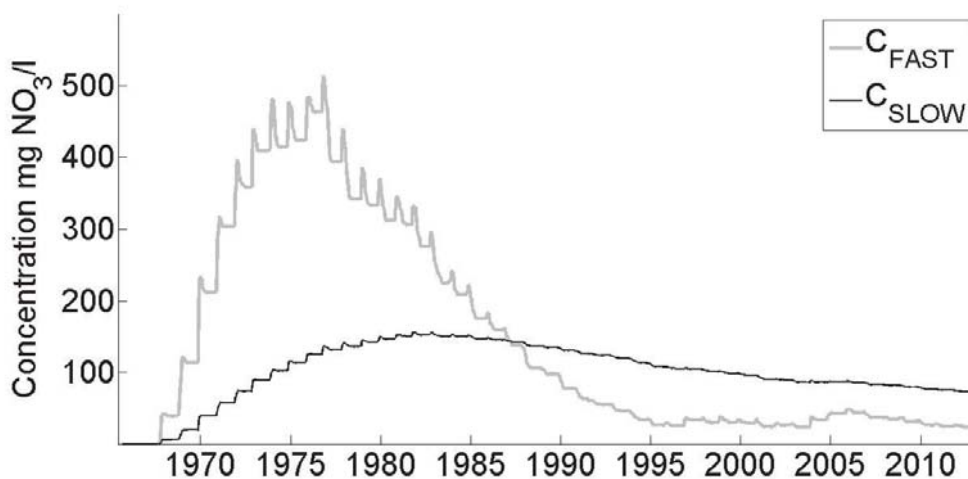
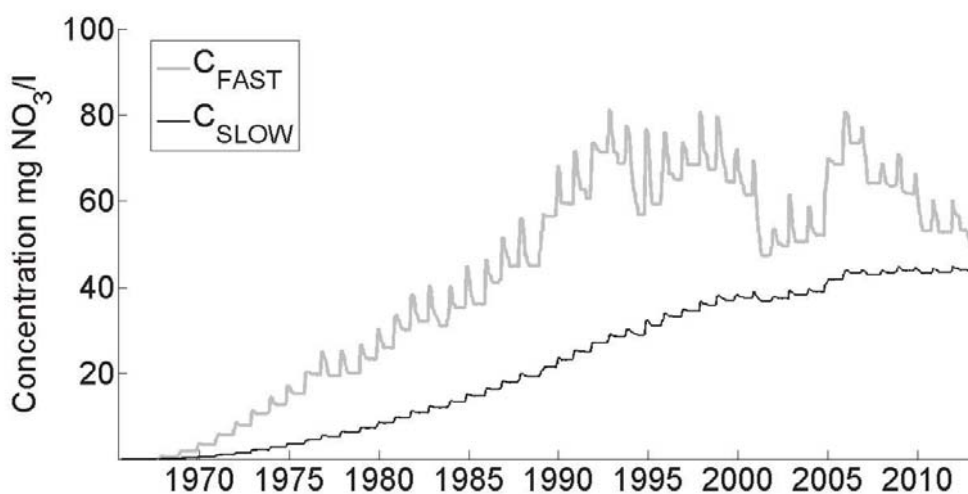
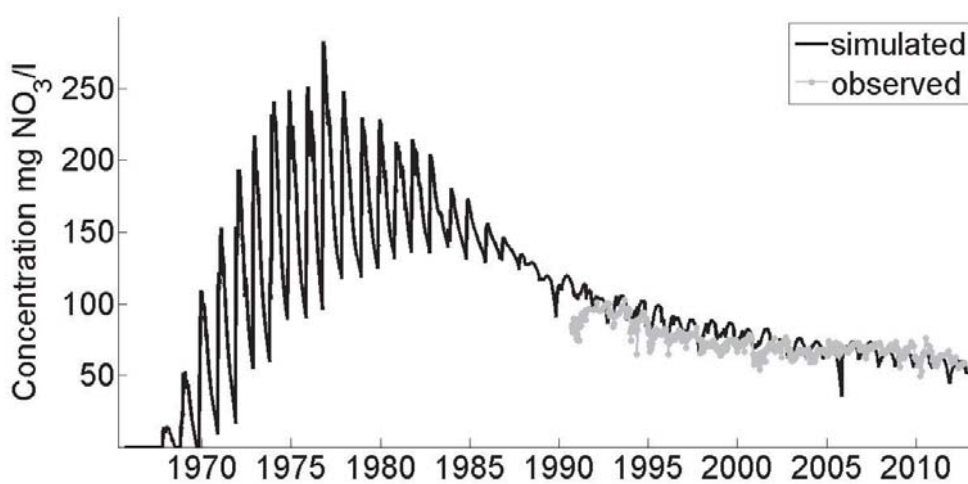


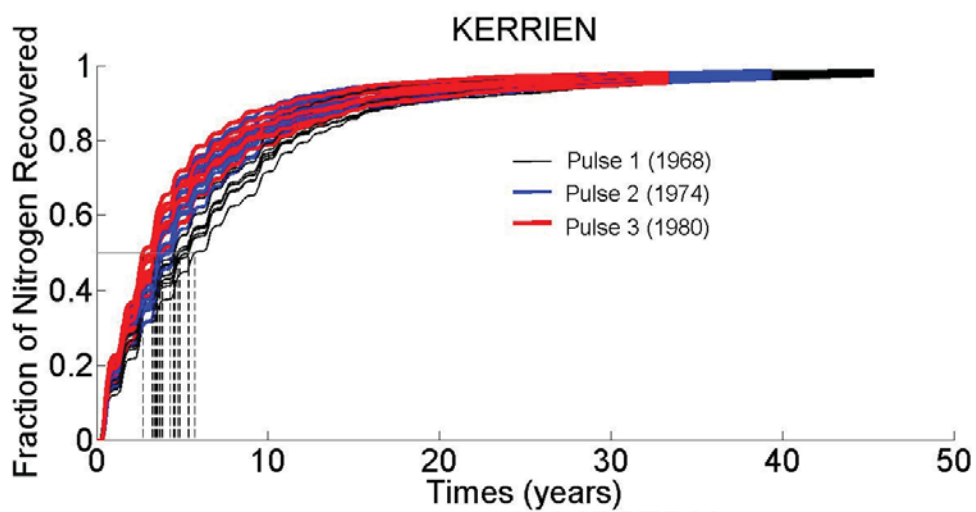
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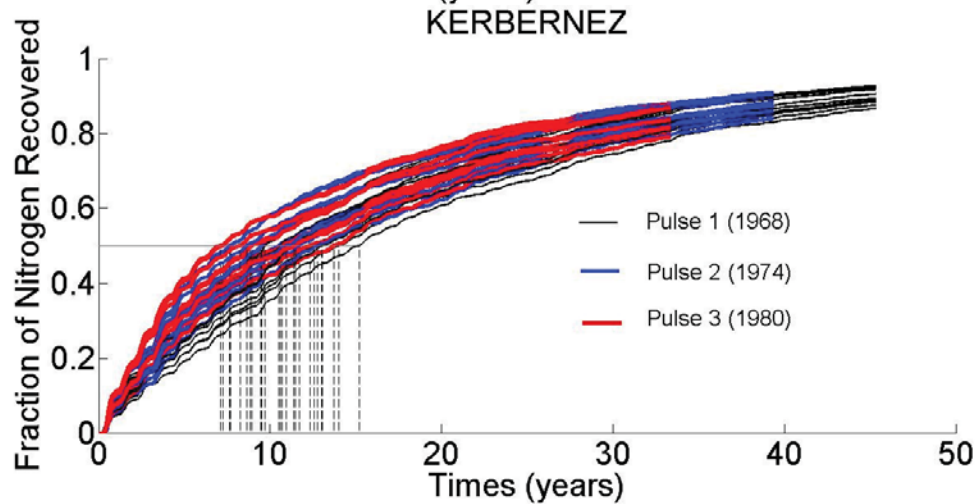
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1 **Highlights**

2

3

- 4
- we used stream nitrate concentration to estimate transit time in catchments
 - 5 • we calibrated a conceptual lumped model on two paired catchments using GLUE
 - 6 • the model simulated reasonably nitrate patterns in stream and groundwater
 - 7 • transit time estimations were consistent with previous studies in the same site
 - 8 • transit time was as sensitive to climate variations as to parameter uncertainty

9

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