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Riverine anaerobic ammonium oxidation across contrasting geologies

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Rivers are an important global sink for excess bioavailable nitrogen: they convert approximately 40% of terrestrial N-runoff per year (~47 Tg) to biologically unavailable N_2 gas and return it to the atmosphere.[1] Currently, riverine N_2 production is conceptualised and modelled as denitrification.[2-4] The contribution of anaerobic ammonium oxidation (or anammox), an alternate pathway of N_2 production important in marine environments, is not well understood.[5,6] Here we use *in situ* and laboratory measurements of anammox activity using ^{15}N tracers and molecular analyses of microbial communities to evaluate anammox in clay, sand, and chalk-dominated river beds in the Hampshire Avon catchment, UK during summer, 2013. Anammox *hzo* gene abundance varied across the contrasting geologies. Anammox rates were similar across geologies but contributed different proportions of N_2 production because of variation in denitrification rates. In spite of requiring anoxic conditions, anammox, most likely coupled to partial nitrification, contributed up to 58% of *in situ* N_2 production in oxic, permeable riverbeds. In contrast, denitrification dominated in low permeability clay-bed rivers, where anammox contributes roughly 7% to the production of N_2 gas. We conclude that anammox can represent an important nitrogen loss pathway in permeable river sediments.

Humans have greatly altered the global nitrogen cycle through industrial N_2 fixation and application of this fixed-N to the land, disturbing the balance between N_2 fixation and N_2 production.[1] Almost half of global terrestrial N_2 production occurs within freshwaters (rivers, lakes, groundwater)[2] which to date has been conceptualised as a simple function of labile organic matter availability, i.e. canonical denitrification.[4,7] Anammox alters the fundamental stoichiometry of the complete mineralisation of organic matter as, for every mole of organic-N converted to N_2, only half of the N-bearing compounds need partial oxidation to nitrite, and for each mole of nitrate/nitrite (NO_χ^-) reduced, one mole of more bioavailable ammonium is also removed.[8] Rivers may not appear the most suitable environments for anammox – labile

carbon,[9] supplied from the catchment, and variable redox environments[10] in the sediments, should, in theory, favour heterotrophic denitrification by facultative anaerobes.[11,12] If active however, anammox alters our perception of how riverbeds function and increases a river's capacity to attenuate nitrogen.

Much of what is known about anammox in the environment comes from estuaries and coastal seas where anammox varies in response to sediment reactivity. The relative contribution of anammox to marine N_2 production (*ra*) decreases with proximity to the shore as supply of carbon stimulates denitrification over anammox.[12,13] Extrapolating this trend further inshore suggested anammox activity would be insignificant in estuaries but anammox potential actually increased.[14,15] In both estuaries and coastal seas, however, anammox is important in low permeability sediments (*ra* <1 to 11 %)[9,16], where oxygen penetration is restricted[12,15] and it is these muddy sediments that the few studies of riverine anammox have occurred.[5,6] In addition, anammox is widespread in marine sediments but the affiliated bacteria are phylogenetically constrained. In contrast, freshwater environments have been shown to possess highest anammox diversity, purportedly containing many novel anammox bacteria.[17]

The geology of the Hampshire Avon catchment (United Kingdom) is dominated by permeable chalk from the Upper Chalk formation underlain by less permeable Upper Greensand and smaller outcrops of impermeable Gault clay (*see* Supplementary Information). Using a combination of *in situ* and laboratory-based ^{15}N tracer techniques[12,18] and molecular assays we characterised both the anammox community and its activity within rivers from clay, sand and chalk-dominated sub-catchments under summer, base flow conditions (Table S1). For rivers in which *in situ* measurements were performed we indexed catchment permeability by calculating the base-flow index (BFI, Table S2), the proportion of river flow derived from deep groundwater sources. In clay catchments, low soil permeability leads to routing of rainfall overland or through shallow, more permeable soils into the river (low BFI). Whilst in chalk or

sand catchments, the higher soil permeability allows infiltrated water to percolate deeper into the aquifer and follow much longer flow paths to towards the river (high BFI).

We began by characterising the anammox *hzo* functional gene that encodes hydrazine oxidoreductase which catalyses the oxidation of hydrazine to N_2. The *hzo* gene was detected in all sediments confirming that anammox bacteria were present (Table S3). Anammox activity was then confirmed by production of $^{29}N_2$ following addition of $^{15}NH_4^+$ and $^{14}NO_3^-$ to anoxic sediment slurries (Table S4). We can attribute this oxidation of ammonium to anammox rather than reduction of metal oxides,[19] for example, as no ^{15}N-N_2 was produced in $^{15}NH_4^+$ only controls (Table S4). Anammox potential varied across the riverine gradient with fastest rates and greatest anammox contribution to N_2 production observed in the permeable sands and chalk-gravels (Figure 1a, *see* Table S5). Anammox potential was also positively correlated with *hzo* gene abundance (r_s (7) = 0.867, p = 0.005). The absolute abundance of the *hzo* gene was significantly higher in chalk-gravels ($F_{(2,6)}$=8.64; p=0.017) and the proportion of *hzo* to 16S rRNA was even greater (Table S3), given that 16S rRNA copies were highest in the clays.

The anammox functional *hzo* gene was sequenced and phylogenetic analysis revealed four clades (Clade I-IV) that differed in their relative distributions between the three geologies. In general, there was a broad diversity of *hzo* sequences that were distinct from known *hzo* sequences (Figure S1a; Table S6; Supplementary discussion 1). In addition, we sequenced the 16S rRNA gene, and clustered 951,000 sequences into 28,000 OTUs. All Planctomycete sequences represented only 0.5-0.9% of the total 16S rRNA sequences, yet none were assigned to anammox genera (RDP classifier), or grouped phylogenetically with any of the currently known anammox bacteria suggesting that anammox bacteria were not detected due to their low relative abundance in the bacterial communities (Table S3).

Somewhat surprisingly, ^{15}N-N_2 production was not limited to anoxic slurries as we also measured ^{15}N-N_2 production after addition of $^{15}NH_4^+$ to slurries with an air headspace (i.e. O_2

saturated; Figure 2, Figure S2). Sediments at the start of the incubation contained considerable $^{14}NH_4^+$ but little $^{14}NO_\chi^-$ (e.g. 350-960 μM and $\leq$3 μM, respectively; Table S7) which, in combination with $^{15}NH_4^+$, could result in $^{29}N_2$ and $^{30}N_2$ through either anammox or denitrification (or both) coupled to nitrification. As production of ^{15}N-N_2 happened immediately upon addition of $^{15}NH_4^+$ (Figure 2), nitrification must be rapid and the coupling to pathway(s) of N_2 production very tight. The potential for aerobic nitrification to fuel anammox has been demonstrated in oceanic waters with no measurable oxygen[20] and inferred in riparian soils.[21] Here we confirmed nitrifications' direct involvement in oxic N_2 production in sediments by addition of allylthiourea, an inhibitor of aerobic ammonium oxidation, which turned off N_2 production completely (Figure 2).

To apportion oxic N_2 production to coupled nitrification-anammox or nitrification-denitrification we modelled the distribution of isotopes within the N_2 produced via either pathway (*see* Methods). Despite clear evidence of nitrification within the slurries (*see* above), the majority of samples (31 of 45 incubations) did not have measurable $^{15}NO_\chi^-$ after the $^{15}NH_4^+$ addition (Table S7) and, therefore, coupling between nitrification and N_2 production was 100 % efficient. Without measurable NO_χ^- we can only assume that nitrification and N_2 production are so closely affiliated that the ^{15}N-content of the NO_χ^- and NH_4^+ pools are equal which, by definition, prevents separation of anammox from denitrification.[22] In the remaining incubations (n = 14; 5 and 9 for chalk-gravel and sand, respectively), $^{15}NO_\chi^-$ was detectable and the ^{15}N-labelling of the NO_χ^- and NH_4^+ pools was different. Production of ^{15}N-N_2 within the $^{15}NO_\chi^-$-bearing subset of incubations was representative of the entire dataset (Figure S2); although no $^{15}NO_\chi^-$ was detected within any clays (n = 15 time series, consisting of 75 discrete sediment samples).

We could apportion the production of N_2 gas to either anammox or denitrification in some of the $^{15}NO_\chi^-$-containing experiments (n = 8 of 14), with 38±2 % and 65±15 % of potential

120 N_2 production occurring via anammox in the chalk-gravels and sands, respectively (mean ± 1
121 s.e; Figure 2). Within the remaining $^{15}NO_\chi^-$-bearing experiments (n = 6 of 14), the N_2 pool was
122 more enriched in ^{15}N than could be explained by either denitrification or anammox (deviation
123 between measured and predicted ^{15}N-content of produced N_2 was 26±4 %, mean ± 1 s.e.; Table
124 S8). This observation violates the assumption of a random combination of isotopes which is
125 fundamental to ^{15}N-assays and, for the N_2 to be more enriched in ^{15}N than predicted, suggests
126 heterogeneity in the ^{15}N labelling of substrate pool(s).[23] Here, the heterogeneity probably exists
127 because the NO_2^- pool actually being reduced is partially physically isolated from the bulk NO_χ^-
128 pool, further supporting a tight coupling of nitrification to N_2 production. Heterogeneity in the
129 NH_4^+ pool is less likely as ammonium was plentiful – both as ambient $^{14}NH_4^+$ (Table S7) and
130 added $^{15}NH_4^+$ (500 μM 98 % ^{15}N).

131 Supply of nitrite rather than ammonium has been suggested as the limiting factor for
132 anammox in aquatic sediments – potentially coupling anammox to either nitrification and/or
133 denitrification.[15] In anoxic marine and estuarine sediments, anammox can be fuelled by
134 denitrification-derived nitrite[15], even forming a symbiotic relationship with some nitrate
135 reducing / sulphur oxidising bacteria.[24] Association between nitrifiers and anammox bacteria
136 may be weak in low permeability sediments (clays, estuarine mud) because much of the nitrite
137 produced in the upper few millimetres of the bed can diffuse into the overlying water or be
138 fully oxidised to nitrate before reaching the sub-oxic layer.[15] In the presence of oxygen
139 however, affinity between nitrifiers and anammox bacteria can exist in aggregates and is indeed
140 the fundamental principle of CANON waste-water treatment reactors operating at reduced
141 oxygen.[25] The ability for anammox bacteria to couple to both aerobic and anaerobic pathways
142 of nitrite production could be very advantageous in permeable riverbeds, where groundwater-
143 surface exchange facilitates advective transport of solutes,[26] creating a mosaic of redox micro-
144 environments[27] within which both nitrification and denitrification can occur.[18]

Rates of *in situ* anammox activity were similar across riverbeds of differing geology (Figure 1b), despite large differences in porewater oxygen – clays were essentially anoxic whilst oxygen was present in both the sand and chalk-gravels (8 to 110 % of air-equilibration; mean ± 1 s.e. = 134±14 μM; Figure S3). In contrast, anammox did make a markedly higher contribution to N_2 production in the permeable sediments compared to the clays (Figure 1b, Table S5); not because anammox activity increased, rather that denitrification activity declined (Table S9). Similar increases in the significance of anammox at the expense of denitrification are well documented in marine sediments.[12] Differences in *ra* between clays, sands and chalk-gravels were consistent across seasons (Figure S5) and are related to the chemical gradient inherent to porewaters of these different riverbed types. Within clays, porewaters were typically reduced (high in ammonium, iron (II) and phosphate) whereas chalk-gravel porewaters were more oxidised (high in nitrate, intermediate in O_2; Figure S4) with *ra* increasing as porewaters become more oxidised (Figure 3a; $r_s(38) = -0.73$, $p<0.001$); hinting at a potential coupling between anammox and nitrification. Anammox activity was strongly associated with nitrite and O_2, increasing as nitrite accumulates in partially oxygenated sediments (Figure 3b) – again mirroring the coupling between partial nitrification and anammox in aggregates as exploited in CANON reactors.[25] In contrast, anammox activity had essentially no association with ammonium, as where ammonium accumulates in these riverine sediments, labile organic carbon (the source of the ammonium) must also be plentiful (Figure 3b), fuelling denitrification at the expense of anammox.[12,13] Overall we found a very strong increase in the contribution of anammox to N_2 production and *hzo* abundance (as a fraction of total bacteria, *hzo*:16S rRNA sequences) with increasing BFI ($r_s(6) = 1.0$, $p = 0.003$ and $r_s(6) = 1.0$, $p < 0.001$ for anammox contribution and *hzo* abundance, respectively; Figure 3c), suggesting, in the long-term, that anammox is favoured with stable conditions (nutrients, temperature, pH).[11]

Here, we have shown how anammox is making a significant contribution to the removal of fixed-N in oxic, permeable riverbeds; a pattern completely at odds with current knowledge. Supply of nitrite to anammox from partial nitrification removes the stoichiometric constraint of denitrification-anammox coupling (*ra* ≤29 %),[28] allowing anammox to potentially be as important an N-sink as denitrification in permeable riverbeds (maximum *in situ ra* = 58 %, median = 37 % for chalk-gravels; Figure 3a). In the clays, anammox proceeds as per muddy estuarine sediments, making only a minor contribution to N_2 production (*ra* ≤7 %) and being fuelled by canonical denitrification.[15] It is important to appreciate that the stoichiometry of anammox, requiring only partial oxidation of some N-substrates, increases the efficiency of rivers to remove fixed N as both NO_χ^- and ammonium, changing our understanding of the ecosystem services they provide.

Data sources

URL and DOI for activity data to be provided once deposition into the Environmental Information Data Centre is complete. *hzo* gene sequences from this study are deposited in GenBank (NCBI) under the accession numbers ---

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Author contributions

MT with CMH, AB and KL conceived the original project. KL performed ^{15}N-related work and with MT interpreted the process data and drafted the original manuscript. BAM designed and performed all the molecular work and phylogenetic analysis. AJD constructed the bioinformatic pipeline and performed the NGS analysis. CW directed the molecular component of project. LO assisted with fieldwork and performed sediment characterisation. AB and CMH performed hydrologic measurements and calculated base-flow indices. All authors contributed to writing the paper and approved the final manuscript.

Competing financial interests

The authors declare no competing financial interests.

Figure Captions

Figure 1: Anammox activity, both rate and contribution to N_2 production (*ra*), differs across a riverine gradient. Activity was measured as total potential in anoxic slurries **(a)** and ambient rates by direct, *in situ* measurements (*see* Supplementary Methods) **(b)**. Grey bars indicate significant differences between groups. Data are mean values ±1 standard error (n = 5 and 10 for **a** and **b**, respectively).

Figure 2: Production of ^{15}N-labelled N_2 following addition of $^{15}NH_4^+$ to chalk-gravels (a) and sand (b) incubated under air-saturated conditions. Notice that in the presence of allylthiourea (white circles), an inhibitor of nitrification, there was no production of ^{15}N-N_2 but without the inhibitor (black circles) there was immediate conversion of $^{15}NH_4^+$ to ^{15}N-N_2 confirming the tight coupling between nitrification and N_2 production. Grey boxes are dissolved O_2. The pathways of N_2 production (anammox or denitrification, red versus white column sections, respectively) were determined via modelling (*see* Supplementary Methods). Data are mean values ±1 standard error (n = 5).

Figure 3: Anammox varies with both patch scale and sub-catchment river characteristics. Differences in anammox contribution to *in situ* N_2 production within clays (red, n=10), sands (blue, n=20) and chalk-gravels (yellow, n=20) result from fine-scale chemical variation **(a)** (lower scores are more oxidised porewaters, *see* Figure S4). Anammox activity is most strongly associated with nitrite and O_2 **(b)**. Squares represent geology averages in the redundancy-analysis triplot. At the sub-catchment scale, both anammox contribution and bacterial abundance (*hzo* copy number: total bacteria) increase markedly with increasing base-flow index **(c)**. Data are means ±1 standard error (n = 3 for *hzo* fractions).

Methods

Measurement of potential anammox activity

Collection of sediment. Sediment was collected from nine rivers in the Hampshire Avon catchment in summer 2013 (19-20 August) under base flow conditions. Rivers were in sub-catchments of predominantly clay, sand or chalk (n = 3 per geology). Full site descriptions are provided as supplementary information. At each river, surficial sediments (<5 cm) were removed from five un-vegetated patches of the main channel by hand with Perspex cores (internal diameter = 9 cm). After sediment disturbed within the core settled, the overlying water was gently decanted and sediment for activity measurements was placed in ziplock bags and stored at ~4 °C.

Preparation of slurries. In the laboratory, each bag of sediment was homogenised by gentle stirring and particles > 9 mm (the internal diameter of the vials) were removed. Sediments were then placed in pre-weighed gas-tight vials (Exetainer, Labco) with replicates from the rivers treated as discrete samples. Slurries were prepared with synthetic river water (*see* supplementary information) in a 1:1 sediment-to-water ratio.

The potential for anammox and denitrification was measured in anoxic slurries[29-31] (oxygen-free N_2 headspace, British Oxygen Company) prepared with de-oxygenated synthetic river water in an anoxic hood (CV24, Belle Technologies). Anoxic slurries were pre-incubated in the dark on an orbital shaker (80 r.p.m., Stuart SSL1) for at least 18 hours to remove any ambient $^{14}NO_{\chi}^{-}$. ^{15}N tracers (100 μL, de-oxygenated) were injected through the septa of the vials in the following combinations: $^{15}NH_4^+$ only, $^{15}NH_4^+$ and $^{14}NO_3^-$ and $^{15}NO_3^-$ only. All ^{15}N-salts were 98 atom % ^{15}N (Sigma-Aldrich). Tracers increased ammonium concentrations by 500 μM and nitrate concentrations to 100 or 300 μM in the clay and permeable sediments, respectively. Each sediment and treatment combination consisted of a reference (no tracer added) and a killed control (100 μL 7M $ZnCl_2$ injected prior to the tracer). The $^{15}NO_3^-$ treatment

consisted of 5 additional slurries per sample which were incubated for 0.5, 1, 2, 3 and 6 h on an orbital shaker (as above). The $^{15}NH_4^+$ and $^{15}NH_4^+$ and $^{14}NO_3^-$ treatments were end point only experiments with 1 additional slurry per sample (i.e. T_{final}) incubated for 6 h. At the end of the incubation period, biological activity was stopped by injection of $ZnCl_2$ (as above) and gas-tight vials were stored upside down until analysis. Once headspace analysis was complete (*see* below) vials were opened and water extracted after centrifugation. Sediment was re-suspended twice with ultrapure water (same volume as aqueous phase of slurry) and the supernatant reserved. Water samples were filtered (0.45 μm polypropylene, Gilson Scientific) into plastic tubes (polypropylene, VWR International) and frozen until analysis (*see* below). The mass of sediment within each vial was then determined after sediments had been dried.

A parallel set of oxic slurries (air headspace) were prepared to investigate nitrification potential by addition of air-equilibrated synthetic river water to sediments on the lab bench with no pre-incubation. Air-equilibrated $^{15}NH_4^+$ tracer was injected through the septa of the vials (as above) and slurries were treated as per the anoxic $^{15}NO_3^-$ treatment. A second experiment was devised to examine if N_2 production observed in the oxic slurries was linked to nitrification (Rivers Ebble and Nadder only, 8/11/2013, sampling procedures as above). In addition to the $^{15}NH_4^+$ treatment, a second treatment containing both $^{15}NH_4^+$ and allylthiourea (concentration in slurry = 100 μM), a nitrification inhibitor, was also included. To determine if oxygen depletion occurred within the oxic slurries a set of scaled-up slurries were prepared in 20 mL gas-tight vials (Chromacol). At each time point the vial was opened and the dissolved O_2 concentration of the slurry was measured by inserting a calibrated, fast response micro-electrode (50 μm, Unisense).

355 **Measurement of in situ anammox activity**

356 Impermeable sediments. Ambient rates of anammox and denitrification were estimated in un-
357 vegetated clays by incubation of $^{15}NO_3^-$ in intact sediment cores[32] (Perspex cores with rubber
358 bungs, internal diameter = 3.4 cm, experiments performed between 03/08 and 10/08/2013).
359 Cores were collected by hand from the River Sem (n= 34) and a tributary of the River Sem
360 (Clay 2 in Table S1, n= 29) and incubated on site in a tank full of river water. The amount of
361 $^{15}NO_3^-$ added and the duration of the incubation varied between cores, ranging from 0.05 to
362 2.5 mL of 78 mM $^{15}NO_3^-$ (98 atom % ^{15}N) and 31 to 252 minutes respectively. The range in
363 $^{15}NO_3^-$ amendments aided separation of anammox from denitrification[32] and different
364 incubation times were used to verify ^{15}N-N_2 production was linear. Following $^{15}NO_3^-$
365 injection into the overlying water column, cores were immediately capped with a bung fitted
366 with a magnetic stirrer and placed in the incubation tank. The overlying water column was
367 gently stirred to prevent stratification and light was excluded from the incubation tank. At the
368 end of the incubation the bung was removed from the core and a water sample was quickly
369 withdrawn with a syringe (polypropylene, BD Plastipak). The core was then homogenised by
370 gentle stirring and decanted into a gas-tight vial (12 mL Exetainer, Labco) which was
371 allowed to overflow before being capped. Biological activity was stopped by injection of
372 100 μL of formaldehyde through the septum. The water sample was then filtered and frozen
373 (as above) until later analysis. Four additional sediment cores were retrieved on each day of
374 fieldwork and a water sample and slurrified sample collected (as above) to determine ambient
375 ^{15}N-N_2 concentrations.

376 Ambient chemistry within the clays was determined on porewaters recovered using
377 rhizon samplers[33] (0.2 μm mesh, 10 cm screen, Rhizosphere) inserted into the riverbed. Rhizon
378 samplers were allowed 24 h to pre-equilibrate[34] before porewater was extracted by applying a
379 vacuum to the rhizon sampler via a syringe held open with a spacer bar. Water samples for

nutrient analysis were processed as described above. Water samples for iron (II) determination were preserved by dispensing porewater directly into a solution of buffered phenanthroline[35] (3.5:1 1M pH 4.5 Acetate buffer: 0.2 % (w/v) 1-10-phenanthroline monohydrate). The dissolved O_2 concentration of the recovered porewater was measured by placing the O_2 microelectrode (as above) into an empty syringe barrel and gently transferring porewater into the vessel with a 2-way valve.[36] We estimate that sample collection and transfer adds approximately 10 μM O_2 to the actual dissolved O_2 concentration and corrected all values accordingly. Following measurement of dissolved O_2 the pH was determined (pH100, VWR International). Additional sediment cores (Perspex, 9 cm diameter) were collected and transported back to the laboratory for fine scale oxygen profiling using a Clark-type oxygen microsensor (OX50, Unisense) within an automated micromanipulator controlled by microprofiling software (SensorTracer PRO, Unisense). Readings from the microelectrode were displayed on a picoammeter (PA 2000; Unisense) and logged after 4 s when the signal had stabilized.

Permeable sediments. Ambient rates of anammox and denitrification were estimated in un-vegetated sediments of the Rivers Ebble, Wylye, Nadder and Avon ("Sand 2" in Table S1) by injection of $^{15}NO_3^-$ into the riverbed and collection of samples over time (i.e. "push-pull" sampling; sampling occurred between 31/07 and 15/08/2013). Ten bespoke stainless steel mini-probes were installed between 4 and 20 cm depth in the bed of the main channel on the day prior to the injection. We modified the system from previous work [36] by attaching an extension (1 m length of 0.75 mm internal diameter Polyetheretherkeytone (PEEK) tubing, Polyflon Technology Ltd.) to the luer connector of the mini-probe to improve speed of sampling. Prior to the injection of $^{15}NO_3^-$, porewater (15 mL total) was withdrawn from each mini-probe for dissolved O_2 and pH measurement, nutrient and iron (II) analysis and natural

abundance ^{15}N-N_2 (as above). Gas samples were collected in 3 mL gas-tight vials and poisoned with $ZnCl_2$ (25 μL, as above).

A tracer solution consisting of 300 μM $^{15}NO_3^-$ (98 atom % ^{15}N) in a synthetic river water/ KCl (4 mM) matrix, was de-oxygenated (as above) and 25 mL aliquots were drawn into luer-lock syringes. Tracer was injected into the riverbed via the mini-probes, with each injection lasting ~20 seconds. Porewater was recovered from each mini-probe immediately post injection and a dissolved gas and water sample was collected (as above). Porewater was then recovered at ~5, 10 and 30 minutes post injection and sampled as above.

Analytical methods for activity measurements

Nitrate (Limit of detection (LOD) 0.4 μM, precision 1 %), nitrite (LOD 0.1 μM, precision 1 %), ammonium (LOD 0.8 μM, precision 3 %) and soluble reactive phosphate (SRP, LOD 0.1 μM, precision 1 %) were quantified by automated colorimetric analysis using standard methods (San++, Skalar). Iron(II) concentrations were quantified on samples preserved with buffered phenanthroline[35] by absorbance measurement at 520 nm on a UV/Visible spectrophotometer (LOD 1 μM, precision 1 %; Evolution 100, Thermo Fisher). The dissolved oxygen electrode was calibrated with a zero solution (0.1 M sodium ascorbate in 0.1 M sodium hydroxide) and 100% air-equilibrated water (laboratory measurements) or river water (field-based measurements), the dissolved O_2 concentration of which was later determined by Winkler titration. Samples for ^{15}N-N_2 quantification that did not contain a headspace were prepared for analysis by addition of helium (commercially pure grade, British Oxygen Company) with a syringe and a two-way valve (0.5 or 2 mL headspaces were added to porewater and slurrified core samples, respectively) and were equilibrated at 22 °C overnight on an orbital shaker (as above). The isotopic composition of N_2 was determined by injection of 50 or 100 μL of headspace (porewater and core/slurry samples, respectively; CombiPAL, CTC Analytics) into

a continuous flow isotope-ratio mass spectrometer (Delta Plus, ThermoFinnigan) and measurement of mass-to-charge ratios 28, 29 and 30. Further details of calibration are provided as supplementary information. Samples for N_2O determination were prepared by withdrawing a sub-sample of the headspace described above (100 μL for porewater samples and 1-10 μL for slurrified core samples) and injecting it into a gas-tight vial containing 2 nmoles of N_2O (prepared by dilution of 100 % N_2O in a N_2 matrix, British Oxygen Company). The entire contents of these vials was swept into a trace-gas pre-concentrator module (Cryo-Focusing, Precon, ThermoFinnigan) and mass-to-charge ratios 44, 45 and 46 were measured on the mass spectrometer described above. Samples for $^{15}NO_\chi^-$ determination were prepared by reduction of nitrate to nitrite with spongy cadmium (modified from ref. 37 - 5 mL of sample and 0.2 mL of 1 M Imidazole were used and samples were incubated for 2 h on an orbital shaker, as above). Samples were then transferred to gas-tight vials (3 mL Exetainer, Labco) and a 0.5 mL helium headspace was added (as above). Nitrite was reduced to N_2 by injection of sulphamic acid through the septa (100 μL 4 mM sulphamic acid in 4 M HCl; B. Thamdrup, personal communication) and, after overnight equilibration, the headspace was analysed for ^{15}N-N_2 as above. The amount of $^{15}NO_\chi^-$ within each vial was determined by preparation of a calibration curve of differing amounts of $^{15}NO_3^-$ (treated as above) versus the mass-to-charge ratio 29: sum of all areas.

Calculations for activity measurements

Production of ^{15}N-N_2, anammox and denitrification potential in anoxic slurries were calculated using standard procedures.[30] Rates of ambient anammox and denitrification were calculated using methods previously applied to intact sediment cores with differences in the ^{15}N-labelling of the N_2 and N_2O pools used to determine the contribution of anammox.[32] In oxic slurries, anammox and denitrification were apportioned by comparing the proportion of ^{15}N in the

454 produced N_2 to anammox and denitrification endmembers in a mixing model. All calculations
455 used to derive rates, contribution of anammox to N_2 production and other parameters (e.g. base
456 flow index) are provided as supplementary information.
457
458 **Statistical methods for activity measurements**
459 All statistics were performed in R[38] (version 3.1.1) using RStudio[39] (version 0.98.1091).
460 Differences in anammox activity between groups was tested with linear mixed effects models
461 using the nlme package[40] where geology or permeability were fitted as fixed effects and site
462 was a random effect.[41] Model fit was improved by adding variance structure to the model
463 allowing variance to differ between groups. Significance of fixed effects ($p < 0.05$) were
464 determined by log likelihood ratio tests between the model of interest and a reduced model,[41]
465 i.e. with no fixed effect but just a random intercept (*see* Table S5).
466 The effect of porewater chemistry on anammox and denitrification was examined using
467 multivariate techniques. First we used principal component analysis (PCA) to investigate
468 correlations between chemical variables and differences in porewater chemistry between rivers.
469 The PCA reduced 7 chemical variables (nitrate, nitrite, ammonium, dissolved O_2, SRP, iron
470 (II) and pH) to two principal components (total variance explained = 74%). Principal
471 component 1 accounted for 56 % of the variance and comprised strong positive loadings for
472 ammonium, Iron (II) and SRP, strong negative loadings for nitrate and pH and an intermediate
473 negative loading for O_2 (Figure S4). We have interpreted this axis as a chemical gradient
474 moving from reduced porewaters, where mineralisation products such as ammonium and SRP
475 accumulate (high scores), to oxidised porewaters (low scores) high in nitrate and intermediate
476 in O_2. The chemistries captured within PC1 separate data into their respective geologies (Figure
477 S4), essentially converting our categorical "gradient" of permeability (i.e. clay, sand or chalk)

into a true riverine gradient. PC2 was most strongly associated with oxygen (positive) and nitrite (negative, Figure S4), however, this axis only explained 19 % of the variance.

We then performed a redundancy analysis using the vegan package[42] with the same chemical dataset, geology as a grouping factor and response variables ambient anammox and denitrification rates and the contribution of anammox to N_2 production (*ra*). We attempted to determine the most parsimonious model by performing stepwise addition of the variables, however, after inclusion of the factor "geology" there were no significant improvements to the model Akaike Information Criterion. Geology alone, i.e. sand or clay, is not very useful for determining chemical controls on riverine anammox but when removed from the model the goodness of fit was considerably reduced (>11% reduction observed). We therefore determined the most parsimonious model by manually comparing adjusted R^2 values following the addition of chemical variables.[43] The explanatory variables of the simplest model were found to be geology, ammonium, nitrate, nitrite and O_2 (Figure 3b, Adjusted $R^2 = 0.40$). In this simplest model 78 % of the variance was explained by the 1st canonical axis which had similar chemical loadings as PC1 in the original PCA.

Relationships between anammox and other variables (e.g. *hzo* gene copy number) were quantified using Spearman's rank correlation on untransformed data with $p<0.05$ used as the criteria for significance.

Molecular analyses

Collection of sediment. Sediment collected for potential anammox activity (see above) was sub-sampled for molecular analysis ($n=3$ for the 9 rivers sampled). Sediment was placed in sterile tubes and preserved cryogenically at -150°C.

qPCR gene abundance. DNA was extracted from 0.25 g wet weight sediment using PowerSoil® DNA Isolation Kit (MO BIO Laboratories, Inc). Gene abundance was quantified

by qPCR with SensiFAST SYBR No-ROX Kit (Bioline) on a CFX96 Real-Time PCR Detection System (BioRad) using the 16S rRNA primer pair Bakt_341F (CCTACGGGNGGCWGCAG) and Bakt_805R (GACTACHVGGGTATCTAATCC)[44] and the *hzo* primer pair HZO-1F (AAGACNTGYCAYTGGGGWAAA) and HZO-1R (GACATACCCATACTKGTRTANACNGT).[45] Gene abundances were quantified by absolute quantification method against an internal standard calibration curve of DNA standards of the target gene from 10^2 to 10^6 copies in 20 µl reactions containing 400 nM of primers and 1 µl of DNA template. Cycle conditions were 95 °C for 2 min followed by 40 cycles at 95 °C for 10s then 60 °C for 30 seconds. Amplification of a single product was confirmed by melting curve analysis.

Amplicon sequencing. Amplicon libraries were prepared by a 28-cycle PCR using primers containing the same target region as the qPCR primers but flanked with Illumina Nextera overhang sequences. Amplicons were purified using AMPure XP (Agencourt) SPRI bead protocols, before adding Illumina flowcell adapter sequences, and one of 96 unique combinations of Nextera paired-end Indexes via a 8-cycle PCR. Amplicons were again purified using AMPure XP beads, quantified using a Quant-iT Picogreen dsDNA assay kit (Life Technologies) on a Nanodrop 3300 fluorospectrometer (Thermo Scientific) and then pooled in equimolar concentrations. The amplicon libraries were quality checked using a DNA 1000 kit on at 2100 Bioanalyzer (Agilent) before sequencing was performed on the Illumina Miseq platform using a MiSeq reagent kit V3 (2 × 300 bp) at TGAC (The Genome Analysis Centre, Norwich). The sequencing reads were analysed using the QIIME pipeline and associated modules.[46] Sequences were de-multiplexed using the Nextera Indexes and quality filtered to remove sequences below Q20 or that contained, any errors in the primer region, above 6 ambiguous bases, and chimeras. The quality filtered reads were clustered into operational taxonomic units (OTUs) using the USEARCH algorithm[47] at the 0.95 level (*hzo*) or 0.97 level

(16S rRNA). 16S rRNA representative sequences from each OTU were assigned taxonomic identities with the RDP classifier.[48] Statistical analysis was performed in the R statistical language version 3.1.3 using the R base libraries[38] and the community ecology analysis-specific package 'vegan'.[42] *hzo* gene multiple sequence alignment was performed on the 100 most abundant OTUs (representing 92-93 % of all sequences in each geology) and codon aligned deduced amino acid seqeunces using MUSCLE (**MU**ltiple **S**equence **C**omparison by **L**og- **E**xpectation)[47] and phylogenies were constructed in MEGA6[49] The nucleotide sequence evolutionary history was inferred by using the Maximum Likelihood method based on the General Time Reversible model.[50] Initial trees for the heuristic search were obtained by applying the Neighbor-Joining method to a matrix of pairwise distances estimated using the Maximum Composite Likelihood (MCL) approach. A discrete Gamma distribution was used to model evolutionary rate differences among sites. The amino acid evolutionary history was inferred by using the Maximum Likelihood method based on the Le and Gascuel 2008 model.[51] Initial tree(s) for the heuristic search were obtained automatically by applying Neighbor-Join and BioNJ algorithms to a matrix of pairwise distances estimated using a JTT model, and then selecting the topology with superior log likelihood value. A discrete Gamma distribution was used to model evolutionary rate differences among sites. Significance of branching order was determined by bootstrap analysis (1000 replicates).[52]

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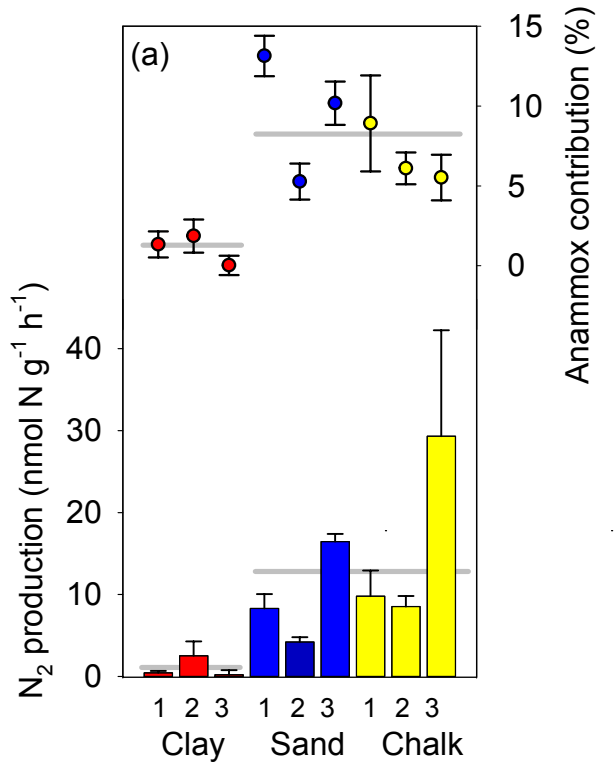
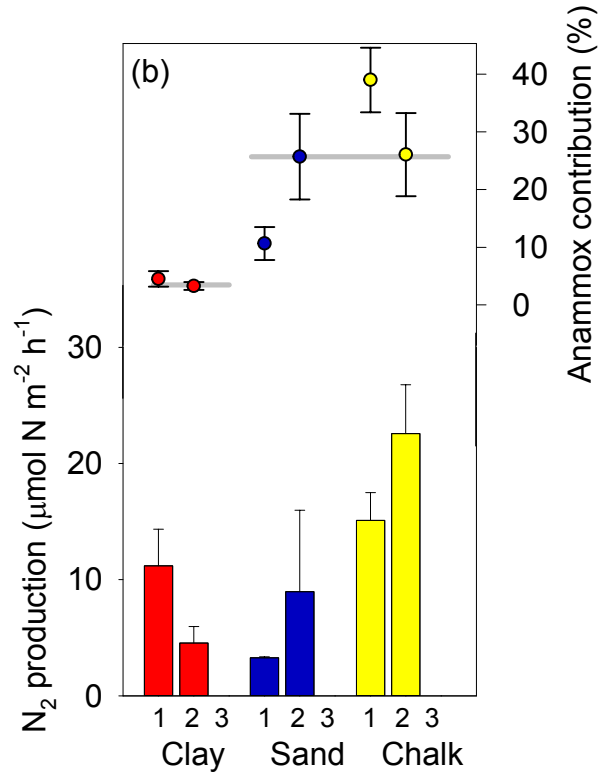
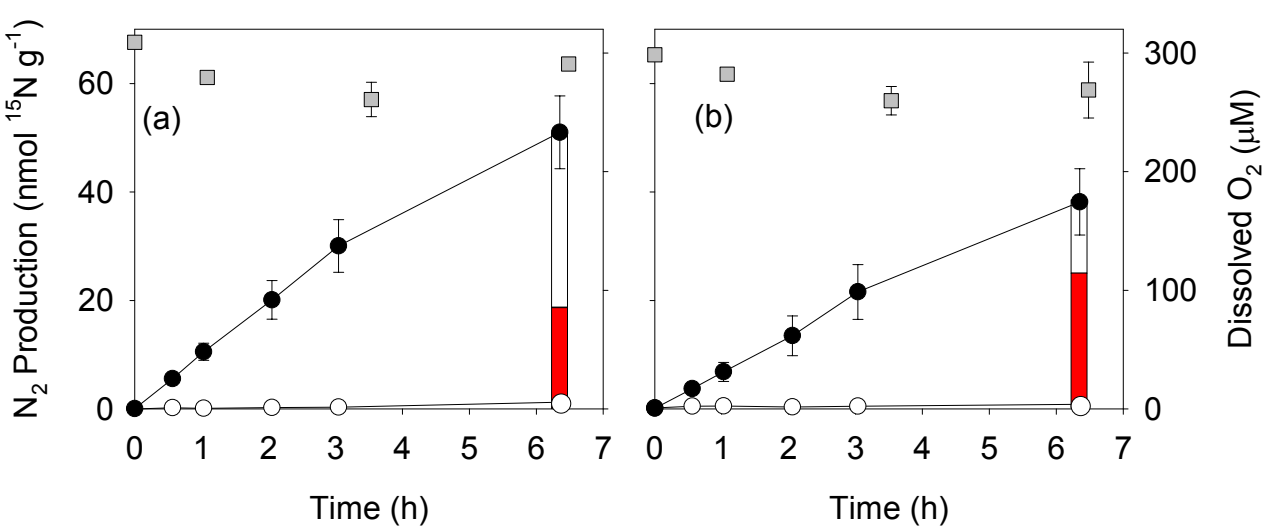
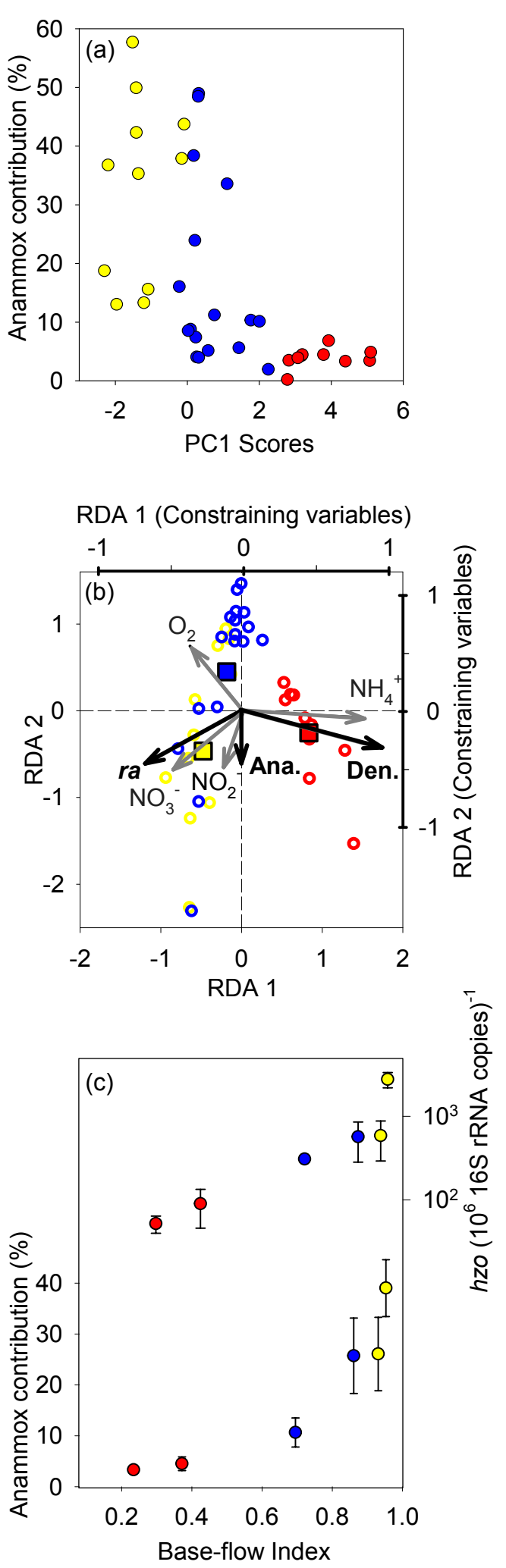

(a)
N_2 production (nmol N g^{-1} h^{-1})
Anammox contribution (%)
0
10
20
30
40
0
5
10
15
1 2 3 1 2 3 1 2 3
Clay
Sand
Chalk


(b)
N_2 production (µmol N m^{-2} h^{-1})
Anammox contribution (%)
0
10
20
30
0
10
20
30
40
1 2 3 1 2 3 1 2 3
Clay
Sand
Chalk


(a)
(b)
N_2 Production (nmol ^{15}N g^{-1})
Dissolved O_2 (μM)
Time (h)
Time (h)
0
20
40
60
100
200
300
0 1 2 3 4 5 6 7


(a)
Anammox contribution (%)
PC1 Scores
(b)
RDA 1 (Constraining variables)
RDA 2
RDA 2 (Constraining variables)
RDA 1
O2
NH4+
ra
Ana.
Den.
NO2
NO3-
(c)
Anammox contribution (%)
hzo (10^6 16S rRNA copies)^-1
Base-flow Index