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TOPICAL REVIEW

# A review of indirect $N_2O$ emission factors from artificial agricultural waters

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## Abstract

Nitrous oxide ($N_2O$) produced from dissolved nitrogen (N) compounds in agricultural runoff water must be accounted for when reporting $N_2O$ budgets from agricultural industries. Constructed ('artificial') water bodies within the farm landscape are the first aquatic systems that receive field N losses, yet emission accounting for these systems remains under-represented in Intergovernmental Panel on Climate Change (IPCC) emission factor (EF) guidelines and global $N_2O$ budgets. Here, we examine the role of artificial waters as indirect sources of agricultural $N_2O$ emissions, identify research gaps, and explore the challenge of predicting these emissions using default EFs. Data from 52 studies reporting dissolved $N_2O$, nitrate ($NO_3$), and EFs were synthesised from the literature and classified into four water groups; subsurface drains, surface drains, irrigation canals, and farm dams. $N_2O$ concentration varied significantly between artificial waters while $NO_3$ did not, suggesting functional differences in the way artificial waters respond to anthropogenic N loading. EFs for the $N_2O$–N:$NO_3$–N concentration ratio were highly skewed and varied up to three orders of magnitude, ranged 0.005%–2.6%, 0.02%–4.4%, 0.03%–1.33%, and 0.04%–0.46% in subsurface drains, surface drains, irrigation canals, and farm dams, respectively. $N_2O$ displayed a non-linear relationship with $NO_3$, where EF decreased exponentially with increasing $NO_3$, demonstrating the inappropriateness of the stationary EF model. We show that the current IPCC EF model tends to overestimate $N_2O$ production in response to $NO_3$ loading across most artificial waters, particularly for farm dams. Given their widespread existence, there is a need to: (a) constrain their global abundance and distribution; (b) include artificial waters in the global $N_2O$ budget, and (c) expand the study of N processing in artificial waters across a geographically diverse area to develop our biogeochemical understanding to the level that has been achieved for rivers and lakes.

## 1. Introduction

Anthropogenic inputs of nitrogen (N) have increased nitrous oxide ($N_2O$) emissions from the world's river networks by four-fold during the 21st century (Yao *et al* 2020). Globally, agriculture contributes ~60% (4.3 Tg N $yr^{-1}$) of anthropogenic $N_2O$ emissions (7.3 Tg N $yr^{-1}$), a result of increasing manure inputs within the pastoral sector and the widespread use of nitrogenous fertilisers on arable land (Tian *et al* 2020). Agriculture contributes to $N_2O$ emissions as a result of the direct production of $N_2O$ within field soils (2.3 Tg N $yr^{-1}$), direct emissions from animal waste and manure management (1.5 Tg N $yr^{-1}$), and from $N_2O$ produced from inland waters (0.5 Tg N $yr^{-1}$) due to increased N loading from agricultural leaching and runoff (Tian *et al* 2020) . The incidental emissions from aquatic ecosystems, termed 'indirect emissions', are recognised as a key component of agricultural $N_2O$ inventories, due to ~24% of terrestrial agricultural N inputs being lost via hydrological pathways (Hergoualc'h *et al* 2019). A large proportion of indirect emissions are generated in water bodies located in close proximity

or connected to agricultural fields, with agricultural drainage waters responsible for 20%–50% of direct $N_2O$ emissions in some agricultural watersheds (Fu *et al* 2018, Billen *et al* 2020).

Emission factors (EFs) applied to dissolved inorganic N measurements are used to estimate $N_2O$ emissions from inland waters where local measurements are lacking. They are used for reporting in both international (i.e. UNFCCC), national, and regional scale contexts to provide bottom-up estimates of greenhouse gas inventories. The Intergovernmental Panel on Climate Change (IPCC) provides guidelines on default EFs for waters contributing to indirect $N_2O$ emissions from leaching and runoff ($EF_5$), which is further subdivided into groundwater and surface drainage ($EF_{5g}$), rivers and reservoirs ($EF_{5r}$), and estuaries ($EF_{5e}$). EFs can be calculated via two different methods using (a) the ratio of the $N_2O$ flux to total N input ($EF_A$), or (b) the ratio of dissolved $N_2O$–N concentration to nitrate ($NO_3$)–N concentration ($EF_B$). Both approaches rely on estimates of N loading to water bodies, which are in turn subject to uncertainties. For aquatic $N_2O$ emissions, previous past assessments to predict global $N_2O$ emissions have focused on rivers, reservoirs, and estuaries (Seitzinger *et al* 2000, Maavara *et al* 2019, Tian *et al* 2020), with little attention paid specifically to artificial surface waters.

The most appropriate EF for each water group has been the centre of controversy for decades, with upwards and downwards revision as more data becomes available. Indirect $N_2O$ EFs from a broad range of water types, including natural and human-made systems, are generalised into the two main groups: $EF_{5g}$ and $EF_{5r}$. Based on a recent revision which derived concentration data from 192 surface waters, the IPCC prescribes a fixed EF value of 0.6% and 0.26% for $EF_{5g}$ and $EF_{5r}$, respectively (Hergoualc'h *et al* 2019). This approach assumes a linear increase in $N_2O$ with $NO_3$ loading, along with assumptions about the rates of $N_2O$ produced from denitrification and nitrification. The range in EFs vary by orders of magnitude for both drainage waters (0.01%–18.4%) and rivers (<0.01%–2.75%, Hergoualc'h *et al* 2019). Poor quantification of indirect $N_2O$ emissions is limited by the lack of studies that incorporate different types of agricultural surface waters (Outram and Hiscock 2012, Xia *et al* 2013, Xiao *et al* 2019), unclear definitions of water types, and inconsistent reporting of factors other than $NO_3$ that can influence $N_2O$ production. Lack of definition for agricultural surface waters assumes that the mechanisms leading to $N_2O$ supersaturation to be similar between groundwater and surface drainage ($N_2O$ source from groundwater), and between rivers, lakes, and reservoirs ($N_2O$ produced *in situ* by nitrification/denitrification). Importantly, current indirect EFs provided by the IPCC do not distinguish between agricultural artificial surface waters, hereafter referred to as 'artificial waters'.

Artificial aquatic ecosystems are designer ecosystems constructed for human purposes, and may be wholly engineered systems where no previous water body existed or physically modified natural waters (i.e. channelized stream) (Clifford and Heffernan 2018). In agricultural landscapes, these artificial waters include channels constructed for subsurface and surface drainage, irrigation canals, and on-farm storage reservoirs (farm dams). These systems are designed to facilitate agricultural production and have human-defined hydrology. Some studies have demonstrated significant differences in dissolved $N_2O$ concentrations between natural and agricultural water bodies, concluding that the application of a single EF value is inappropriate for upscaling indirect aquatic emissions (Outram and Hiscock 2012, Xiao *et al* 2019). Although there has been an increase in studies measuring $N_2O$ and N loading in artificial waters, there remains a lack of quantitative understanding to guide predictions of indirect $N_2O$ emissions from these systems.

Currently, IPCC EFs for estimating indirect $N_2O$ emissions do not distinguish between artificial waters and natural surface waters (rivers, streams, and lakes) in agricultural landscapes. Agricultural surface waters such as drains, ditches, and canals are assigned the same EF as groundwater and headwater streams, yet represent 23% of data determining the default emission value of $EF_{5g}$ (Tian *et al* 2019). Farm dams and other small farm-scale impoundments fall under $EF_{5r}$ for rivers, lakes, and reservoirs, yet only represent ~7% of this category (Tian *et al* 2019). This review aims to address these gaps by: (a) explicitly assessing the role of different artificial agricultural waters as sources of indirect agricultural $N_2O$ emissions; and (b) quantitatively evaluating their response to $NO_3$ loading in the context of the current IPCC indirect $N_2O$ EF estimates. We draw on available data from the published literature on artificial waters and review the knowledge of $N_2O$ production for each water type. We anticipate the findings in this review will lead to greater recognition, improved definition, and eventually, explicit inclusion of artificial waters in agricultural $N_2O$ budgets to help account for uncertainties in agricultural $N_2O$ accounting.

## 2. Methods

### 2.1. Literature search and classification

We employed a similar approach to that of Tian *et al* (2019) in our literature search for studies reporting dissolved $N_2O$ and $NO_3$ concentrations in agricultural surface waters. The inclusion criteria was redefined to only include constructed, engineered, or modified water types located directly within the agricultural landscape (i.e. no rivers, natural creeks, or hydroelectric reservoirs). Data for EFs were gathered from studies included in the most recent IPCC update on $EF_{5g}$ and $EF_{5r}$ (Tian *et al* 2019), and a

**Table 1.** Types of artificial or modified agricultural surface waters and their description considered in this review.

| Agricultural surface water type | Description | Water types included |
|---|---|---|
| Subsurface drains | Subsurface drainage or other drainage structures not exposed to the atmosphere. High $N_2O$ concentrations can accumulate and result in rapid degassing to atmosphere at field outlets. | Tile drains, piped drains, tile drain outlets. |
| Surface drains | Drainage channels for transporting excess agricultural water off the landscape. These structures are exposed to the atmosphere. Viewed as 'zero-order' streams. | Drainage ditches, drainage canals, drainage channels. |
| Canals | Canals and channels transporting irrigation water. Used for both water supply and removal on irrigated fields. | Any canal or channel supplying or removing water on irrigation land. |
| On-farm dams | Small waterbodies (<10 000 $m^2$ area) of open water located directly within the agricultural landscape. Used for water supply, water storage, recreation, and flood control. Aquaculture ponds not included. | Artificial/modified wetlands, treatment wetlands (e.g. dairy effluent), rice paddy wetlands, livestock watering dams, irrigation storage dams, small reservoirs (not used for energy production). |

Google Scholar search was conducted using the key terms '$EF_5$' or 'dissolved $N_2O$ concentrations' combined with water definitions ('wetland', 'pond', 'dam', 'reservoir', 'drainage', 'storage', 'canal', 'ditch', 'tile drain', 'channel') under the theme 'agriculture'. To be included, studies needed to (a) adequately define the water type to ensure it was an artificial or hydrologically modified system; (b) be located within an agriculturally dominant catchment; and (c) have an EF calculated from the $N_2O$–$N/NO_3$–N concentration ratio. We chose to focus on the $N_2O$–$N/NO_3$–N mass ratio method for calculating EFs as most studies lack the detailed mass balance information of N transport and $N_2O$ flux measurement in their respective catchments required for $EF_A$. Mean values from both temporal and spatial datasets were included if they reported $N_2O$ and $NO_3$ concentrations. We opted not to include aquaculture ponds and flooded rice paddy waters as $N_2O$ emissions from these systems would be considered direct agricultural emissions. Following this criteria, an additional 25 studies were collected on top of the 27 artificial water studies included in the current IPCC EF estimates (Tian *et al* 2019).

Table 1 describes the classification of agricultural surface waters considered in this review. Subsurface drains are drainage structures installed directly under agricultural fields to control groundwater levels and prevent waterlogging (Waller and Yitayew 2015). Originally known as tile drains, these structures often consist of a dense network of perforated pipes placed 1–2 m below the surface and which discharge into open collection drains. The concentration of nutrients is highest in these waters as they increase soil water infiltration and collect leached water directly from the soil surface. Surface drains are open waterways, such as ditches, that artificially remove excess water off the farm landscape. They include both on-farm zero-order field drains, main collection drains, and outlets. This infrastructure exists in flood prone areas such as wetlands, floodplains, and poorly draining soils, and acts to enhance agricultural productivity through improved efficiency, timing, and variety of farming operations (Christen *et al* 2001, Herzon and Helenius 2008). Canals represent engineered channel structures that can act as either supply or drainage channels for irrigated agricultural fields. These networks are often constructed in dryland agricultural regions and support intensive irrigated broad acre production of row crops and horticulture. Although drainage ditches and irrigation canals are structurally similar, being linear channels transporting flowing water, we decided to make a distinction here, largely because they represent inherently different farming systems (e.g. ditches located on farms where excess water is a limitation and canals located in dry regions where water needs to be artificially supplied).

On-farm dams are an integral part of the agricultural landscape that exist to store, control, recycle, and treat water on the farm. Here, we include all constructed waterbodies located directly in the farming landscape, including ponds and wetlands that have been modified or impacted by agriculture beyond their natural state. They exist across all areas of agriculture, including arable to livestock land and intensive commercial scale to small landholder scale (Chumchal *et al* 2016). In terms of nutrient cycling these water bodies behave functionally differently to drainage waters (as discussed below) and have been suggested as a management strategy to reduce surface N runoff (Siegfried *et al* 1994, Passy *et al* 2012).

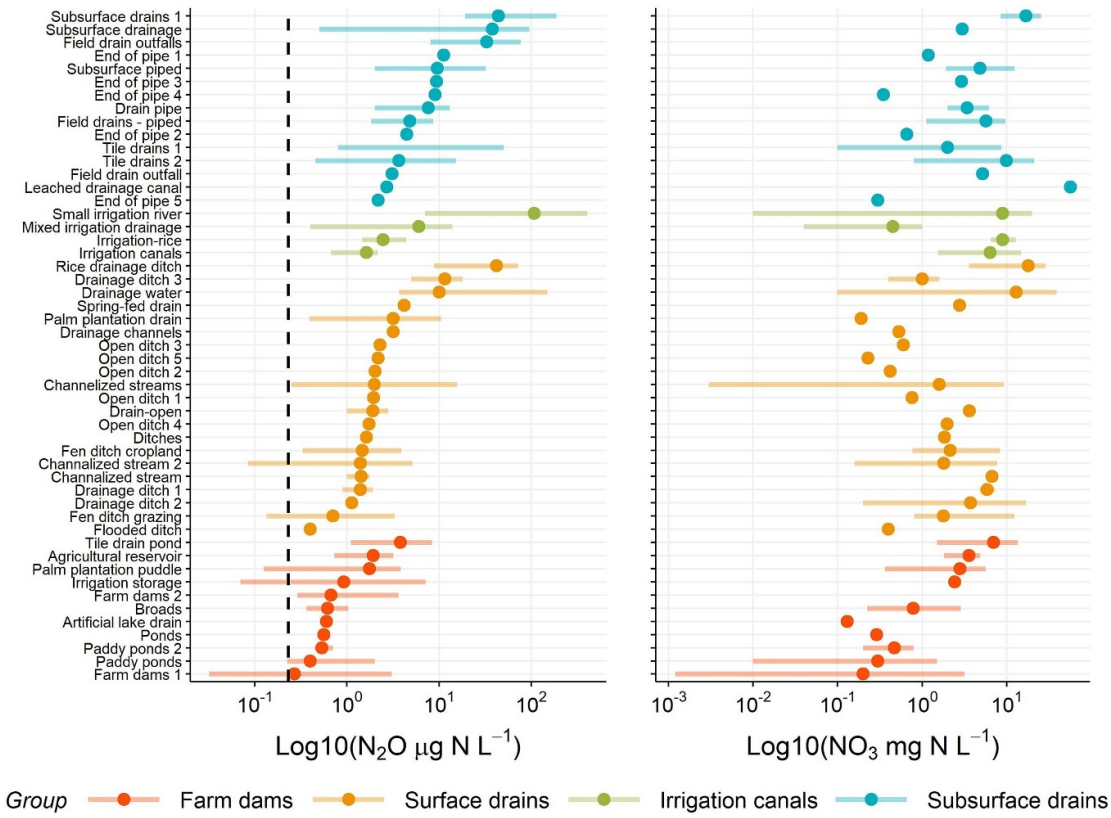


**Figure 1.** Summary of dissolved $N_2O$ and $NO_3$ concentrations from studies compiled in this review on $\log_{10}$ scale. Dots represents means and lines represent range. Dashed line represents dissolved $N_2O$ concentration at atmospheric equilibrium.

### 2.2. Analysis

Data were categorised into four water groups: sub-surface drains, surface drains, canals, and farm dams (table 1). To assess significant differences between water groups, means of $N_2O$, $NO_3$, and EFs were analysed using a one-way analysis of variance followed by a Tukey post-hoc test ($p < 0.05$) performed in R version 3.6.3 (R Core Team 2020). $N_2O$, $NO_3$, and EF data were $\log_{10}$-transformed prior to analysis after checking for skewness using the skewness function in R package e1071 (v1.7–3; Meyer *et al* 2019). EFs were converted to a percentage for plotting and represent the percentage of $N_2O$–N mg $l^{-1}$ relative to $NO_3$–N mg $l^{-1}$ present in the water.

To test the relationship between $N_2O$ and $NO_3$, a Pearson's least square regression for linear covariance ($p < 0.05$), was applied to $\log_{10}$-transformed data and assessed by water group. Models were compared with IPCC predictions by applying $EF_{5g}$ and $EF_{5r}$ (0.0060 and 0.0026) to $NO_3$ data collected in this review, and which assumes a linear increase in $N_2O$ concentrations with increasing $NO_3$. Other environmental drivers that may be of importance in predicting aquatic $N_2O$ emissions were also tested, given data availability. Variables included dissolved organic carbon (DOC), pH, annual N fertiliser applied, and precipitation.

Finally, artificial waters were compared with natural waters to assess whether artificial waters behave differently as sources of indirect $N_2O$ emissions. Here, $N_2O$, $NO_3$ and EF data for natural waters collected in Tian *et al* (2019), for the 2019 IPCC refinement, were used, and included waters defined as groundwater, streams, rivers, and lakes in agricultural catchments.

## 3. Results

A total of 52 studies met the selection criteria for artificial waters defined in this review, which included 15 sub-surface drains, 22 surface drains, 11 farm dams, and 4 irrigation canals. Over half of studies were located in Europe, followed by 30% in Eastern to South-Eastern Asia, 13% in North America, and 8% in the Southern Hemisphere (figure S1 (available online at stacks.iop.org/ERL/16/043005/mmedia)). These included land uses such as arable systems which were mostly annual row-crops ($n = 19$), pasture grazing for dairy and animal production ($n = 18$), rice ($n = 6$), irrigated land ($n = 3$), palm oil plantations ($n = 2$), and a mix of cropping and pasture ($n = 3$).

Mean $N_2O$, $NO_3$ concentrations, and EFs across all studies ranged 0.27–108 $\mu$g $N_2O$–N $l^{-1}$, 0.13–57 mg $NO_3$–N $l^{-1}$, and 0.005%–4.37%, respectively (table S1, figure 1). In five studies, undersaturated $N_2O$ concentrations relative to atmospheric equilibrium were reported, indicating potential negative emissions (figure 1). Within study variation

was large, with up to one and two orders of magnitude between the minimum and maximum recorded values for $N_2O$ and $NO_3$ concentrations, respectively. The largest variation observed was in drainage water from a heavy clay soil (Dowdell *et al* 1979), a small irrigation river for rice (Hasegawa *et al* 2000), and subsurface drainage from an onion field (Sawamoto *et al* 2003). This variation was attributed primarily to seasonal changes. For EFs, within study variation was as low as 0.04% in agricultural reservoirs holding water for irrigation (Wang *et al* 2017) and as high as 38% in a spatial survey of farm dams (Webb *et al* 2019a).

### 3.1. Differences between water groups

Significant differences between artificial waters were only observed for $N_2O$ concentrations (figure 2). Sub-surface drains, surface drains, and farm dams were significantly different from each other with means of 13.04, 6.16, and 1.09 $\mu$g N $l^{-1}$, respectively for $N_2O$ (table S1). Irrigation canals were significantly different from farm dams and had the highest mean $N_2O$ concentration of 29.52 $\mu$g N $l^{-1}$, although this value was highly skewed by one high value (108 $\mu$g N $l^{-1}$). In contrast to $N_2O$ concentrations, $NO_3$ and EFs did not vary significantly with water group. The highest mean $NO_3$ concentration was recorded in a sub-surface drainage site of a vegetable farm, in a region with high annual rainfall (Saitama Prefecture, Japan), while the lowest mean concentration was observed in an artificial lake collecting subsurface and surface drain discharge from dairy farms. In contrast, the subsurface drainage site with the highest $NO_3$ loading reported the lowest mean EF of 0.005% (Hasegawa *et al* 2000).

### 3.2. $N_2O$ and EF relationship with $NO_3$

Across all artificial waters, significant linear relationships for both log-transformed $N_2O$ concentrations ($R^2 = 0.22$, $p < 0.05$) and EFs ($R^2 = 0.29$, $p < 0.05$) were observed with log-transformed $NO_3$ concentrations (figure 3). The rate of increase in $N_2O$ concentrations with increasing $NO_3$ concentration was lower than that predicted by the IPCC $EF_{5g}$ and $EF_{5r}$ models (figure 3(A)). EFs demonstrated an inverse relationship with increasing $NO_3$ loading (figure 3(B)). When considering water groups separately, different responses to $NO_3$ were shown (figure S2), with only farm dams found to have a significant positive correlation between $N_2O$ and $NO_3$ ($R^2 = 0.79$, $p < 0.05$). Subsurface drains, surface drains, and farm dams all had significant inverse relationships between EF and $NO_3$ (figure S3). Relationships were difficult to assess for irrigation canals due to the lack of observations ($n = 4$).

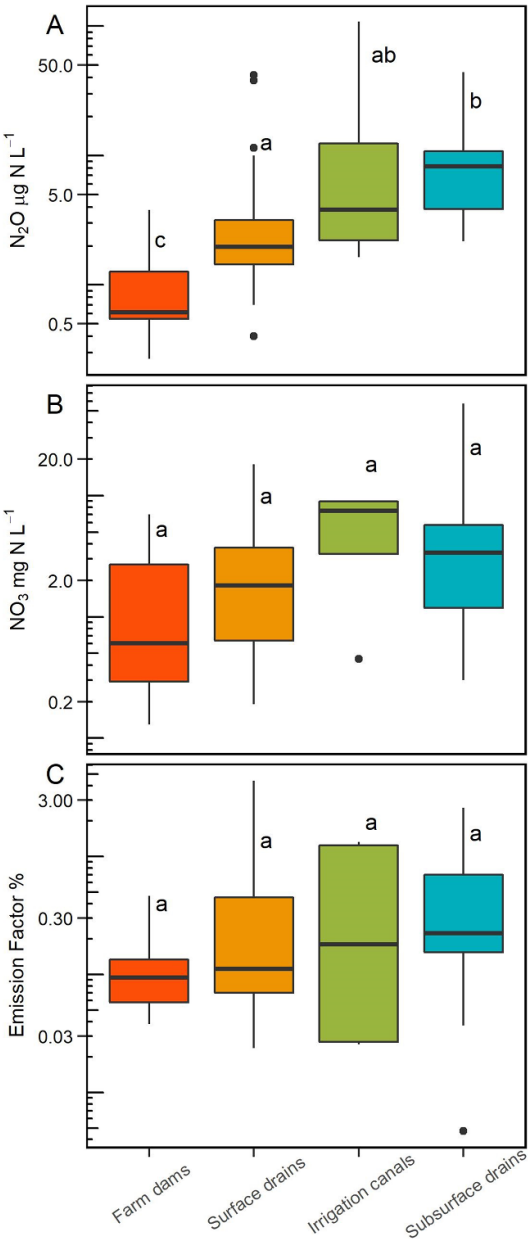


**Figure 2.** Variation of $\log_{10}$-transformed dissolved $N_2O$ concentration ($\mu$g N $l^{-1}$), $NO_3$ (mg N $l^{-1}$) concentrations, and EFs (%) between artificial agricultural surface waters. Different letters represent significant differences ($p < 0.05$) among groups (one-way ANOVA and Tukey–Kramer post hoc test). Boxes span the interquartile range (25%–75% quartiles), whiskers 5%–95% of observations, horizontal line the median, and circle points represent outliers.

### 3.3. Relationships between $N_2O$ concentration and other factors

Regression analysis with log-transformed mean $N_2O$ concentrations revealed significant relationships with the DOC:$NO_3$ ratio, pH, and annual precipitation (figure S4), although these were limited by the number of studies that reported them. Lower $N_2O$ concentrations were observed with either an increase in the DOC:$NO_3$ ratio ($R^2 = 0.2$, $p < 0.05$) or water pH ($R^2 = 0.24$, $p < 0.05$), while a weaker correlation ($R^2 = 0.07$, $p < 0.05$) with annual precipitation

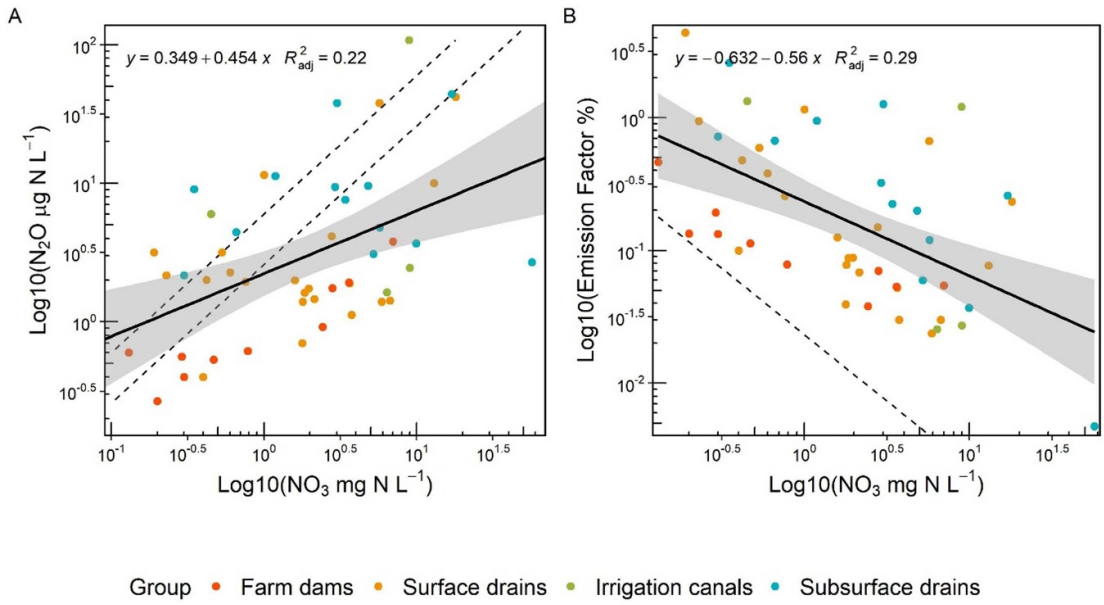


**Figure 3.** Linear regression ($p < 0.05$) of $\log_{10}$ transformed mean dissolved $N_2O$ concentration ($\mu$g N $l^{-1}$) and EFs (%) versus mean $NO_3$ (mg N $l^{-1}$) concentration across studies. Dashed lines in (A) represents the IPCC $EF_{5g}$ and $EF_{5r}$ model with current EF of 0.0060 and 0.0026, and (B) defines $N_2O$ positive (right side) or negative (left side) emissions.

showed an increase in $N_2O$ with higher precipitation. No significant relationship was found between $N_2O$ concentrations and N inputs from fertilizer and manure.

## 4. Discussion

Significant differences in $N_2O$ concentrations were observed between artificial waters, yet the large variability in $NO_3$ concentrations resulted in no distinct difference in EFs (figure 2). Seasonal variability presents a limitation in the current dataset analysis, with some studies reporting temporally resolved means and while others account for spatial variability (e.g. Webb *et al* 2019a), which could influence the lack of difference observed between some water types here. Given only ~50% of the dataset included seasonal measurements, future studies should examine seasonal impacts on EFs, as water temperatures affects gas dissolution and rainfall/drought events can influence $NO_3$ and substrate loadings (White *et al* 2021).

As reported in other studies and in the current dataset, $N_2O$ concentrations do not linearly increase with increasing $NO_3$ concentration, which is the assumption taken when using stationary EFs for estimating indirect emission. This assumption does not take into account the different conditions that drive rates of $N_2O$ production and consumption within each water type. The significant differences in $N_2O$ but not $NO_3$ concentrations observed between subsurface drainage, surface channels, and farm dams supports the idea of mechanistic differences in $N_2O$ processes. Here, we summarise the data and current understanding of mechanisms driving $N_2O$ production in each artificial water type explored in this review.

### 4.1. Subsurface drains

Sub-surface drainage is used to protect soils from waterlogging and manage salinity in agriculture, and in some regions can occur across 80% of the cultivated landscape, often for annual row-crop production and perennial tree crops (Dinnes *et al* 2002). Subsurface drains exist in tandem with surface drains as part of an integrated drainage system, including piped drains, field ditches, collection ditches, and an outlet drain. The global extent of subsurface drainage is unknown, however, it occurs in 10%–100% of agricultural soils across European countries and ~25% in Northern America (Herzon and Helenius 2008). In this review, 15 studies had reported sub-surface drainage $N_2O$ concentration measurements, installed at depths ranging from 0.5 to 1.6 m below the surface. Livestock agriculture was the main land use reported in addition to some mixed arable production. The majority of sites were located in the UK ($n = 9$), with the remaining from Japan (3), Canada (2), and Denmark (1) (figure S1). In almost all the studies, water was collected at the outlet of the subsurface drainage systems and, as a result, likely underestimated the true EF value due to rapid degassing of $N_2O$. This is one of the major limitations of sampling subsurface field drainage and requires consideration in future measurements (Roper *et al* 2013).

The mean EF of 0.52% for subsurface drains was significantly higher than other artificial water groups, and is comparable to the IPCC $EF_{5g}$ recommendation of 0.60% for groundwater and drainage waters. The

higher $N_2O$ concentrations observed within subsurface drains may be attributed to a number of factors (figure 2). Firstly, the key difference between these artificial waters with the others reviewed here is their lack of exposure to the open atmosphere, which regulates the rate of $N_2O$ evolution from the water surface. Additionally, *in situ* production and consumption of $N_2O$ is limited due to both the rapid delivery of drainage water once field water enters the drainage system and lack of bottom substrate to support microbial activity. If tile drains do develop any biofilm area, short water residence times may preclude any significant $NO_3$ removal. Furthermore, supersaturated $N_2O$ concentrations almost certainly originate from the same water source as shallow groundwater: leached soil pore water (Dowdell *et al* 1979). Therefore, subsurface drainage likely represents more of a passive pipe system, transporting dissolved $N_2O$ produced in soil with little outgassing until water reaches a drainage outlet.

$N_2O$ concentrations in subsurface drains are more closely linked to terrestrial characteristics, including soil type, land use and fertilizer practices. A UK study of an intensive arable catchment found that soil texture significantly influenced $N_2O$ concentrations and the relationship with other water quality parameters in a subsurface field drain (Hama-Aziz *et al* 2017). Soils with poor drainage may provide more favourable conditions for $N_2O$ production through increased anoxia and contact with the soil matrix (Hénault *et al* 2012, Jamali *et al* 2016), allowing for more $N_2O$ to dissolve into leached soil water. The strong link with terrestrial N cycling processes is further supported by various studies demonstrating how subsurface drainage design can significantly influence soil N attenuation and direct $N_2O$ emissions (Clagnan *et al* 2018).

### 4.2. Surface drains

Globally, artificial drainage exists in 130–200 million ha of cropland worldwide, and is increasing in extent and intensity (Schultz and De Wrachien 2002, Castellano *et al* 2019). Surface drains are part of the greater artificial drainage system, often co-existing with subsurface drains, yet are responsible for delivering eutrophying nutrients to natural aquatic and coastal ecosystems (Blann *et al* 2009). Here, 22 studies reported $N_2O$–$N/NO_3$–N concentration and EFs in surface drains, with water depths ranging from 0.06 to 0.6 m. Mixed arable production was the predominant land use followed by livestock, rice, and a palm oil plantation (supplementary data 1). Around 50% of studies were located in the UK, with others distributed across Europe, China, Japan, Indonesia, US, and New Zealand (figure S1). Mean EF was 0.46% (0.02%–4.37%), which is lower than the IPCC $EF_{5g}$ recommendation of 0.60% for groundwater and drainage waters.

Drain position and location within the network are likely important variables to consider when estimating $N_2O$ EFs. Surface drains had significantly lower $N_2O$ concentrations compared to subsurface drains, which supports observations in studies measuring surface drains and tile drains in connected artificial drainage systems (Sawamoto *et al* 2002, Reay *et al* 2004a). Variable physical gas transfer, discharge rate, dilution effects, and distance from point sources can substantially influence within-drain $N_2O$ variability (Sawamoto *et al* 2002, Reay *et al* 2004a, Premaratne *et al* 2017). For example, open surface drains can support large atmospheric $N_2O$ losses during the transport of field water, with dissolved $N_2O$ reductions up to 80% (Reay *et al* 2003, 2004b). This is a result of the relatively high gas transfer velocity generated by field drain outfalls, weirs, and changes in discharge velocity based on the size or shape of a drain. As such, large changes in $N_2O$ concentrations relative to $NO_3$ concentrations can occur over small spatial scales which leads to large variability in EFs.

Adding to their complexity, some surface drains may receive water from a variety of land uses and sources, or have a more established benthic structure or vegetation environment dependent on stream hierarchy and channel morphology. Headwater agricultural channels can have exponentially higher $N_2O$ emissions than higher-order waterways due to higher N loading, higher N processing rates, and less water volume dilution effects (Turner *et al* 2015). Headwater channels typically receive substantial sediment yields from agricultural fields, resulting in sediment storage with accumulation rates as high as ~90 kg $m^{-1}$ $yr^{-1}$ (Lecce *et al* 2006). This benthic environment can help attenuate agricultural nutrient loads through denitrification at the sediment-water interface, subsequently removing $NO_3$ from the water column (Veraart *et al* 2017). The current assumption for these drainage systems within the IPCC $EF_{5g}$ model is that $NO_3$ and $N_2O$ are sourced from groundwater with little in-stream nitrification–denitrification. However, studies have shown strong denitrification activity in drainage ditches across a variety of land use and soil types (Veraart *et al* 2017, Soana *et al* 2019).

### 4.3. Irrigation canals

Globally, irrigation covers an area of over 300 million ha, which has doubled in the last 50 years and is expected to expand in developing countries (Schultz and De Wrachien 2002, Neumann *et al* 2011). Irrigation farming systems have some of the largest agricultural N losses from hydrological pathways, up to 46%–76% of total N applied to fields (Thorburn *et al* 2011, Perego *et al* 2012). Therefore, there is a need to characterise $N_2O$ emissions from these dynamic systems, however, findings from this review reveal a severe lack of empirical data in this area.

Of the artificial waters collected in this review, only four recorded dissolved $N_2O$ measurements within irrigation waters (table S1). The studies were distributed in the lower latitude regions including Japan, Australia, and Mexico where cotton, rice, and mixed livestock-rice were the main production types. Reported water depths ranged from 0.03 to 1 m. Irrigated farming systems are most prominent in semi-arid regions, with two thirds located in Asia (Molden 2007). Irrigation canals had a mean EF of 0.65% (0.03%–1.33%), which is comparable to the IPCC $EF_{5g}$ recommendation of 0.60%.

However, with such a limited number of studies, it remains difficult to draw conclusions on the difference in $N_2O$ production from irrigation waters with other artificial waters. A large $N_2O$ concentration range of 1.6–108 $\mu$g N $l^{-1}$ in canals led to no significant difference when compared to subsurface and surface drains (figure 2). Although not shown here, irrigation canals have some key functional differences with surface drainage channels that may give rise to different $N_2O$ emission estimates if the database for these systems increases. Field runoff collected by canals is largely driven by irrigation events rather than rainfall (Harrison *et al* 2005). Further, irrigation canals can sometimes be free of both macrobenthic organisms and benthic macrophytes due to regular maintenance and concrete construction in some high value farming regions (Kitamura and Nakaya 2010). Finally, many on-farm irrigation canals undergo regular wetting and drying cycles with the irrigation regime, or remain flooded but have highly fluctuating water tables, which exposes bottom sediments or channel banks to the atmosphere. The indirect $N_2O$ emissions released from the exposed sediment phase remains an area needing further investigation (Schwenke *et al* 2020).

The timing and length of irrigation events controls the supply of N from soils to the irrigation water and this will influence $N_2O$ levels within irrigation canals collecting irrigation drainage. For example, an Australian cotton farm irrigation study speculated that low $N_2O$ concentrations and EFs observed within the on-farm irrigation channels were due to a soil water deficit in preceding irrigation events and the short surface water application time (Macdonald *et al* 2016). In addition, field water contact with groundwater may further alter the ratio of $N_2O$ to $NO_3$. Comparing two opposing studies reporting either no groundwater contact (Macdonald *et al* 2016) or contact with groundwater (Hasegawa *et al* 2000), the mean EF is two orders of magnitude higher in the groundwater-fed irrigation system. The mixed livestock and rice farm in Japan was an extreme case in terms of $N_2O$ concentrations (7–407 $\mu$g $N_2O$–N $l^{-1}$) within the irrigation group. In this case, high $NO_3$ groundwater from stock breeding areas flowed from altitude into rice paddies with high organic matter and fertiliser N inputs, which stimulated excessive $N_2O$ production via denitrification in the receiving irrigation canal (Hasegawa *et al* 2000).

### 4.4. Farm dams

On-farm dams are ubiquitous within the agricultural landscape, with several million across the globe used for water supply, irrigation, runoff control, and wastewater management (Verstraeten and Poesen 2000). They are highly abundant in agricultural regions within China, the North American Great Plains, and Australia, where densities often exceed 5 per $km^2$ (Renwick *et al* 2006, Grinham *et al* 2018, Chen *et al* 2019). Although agricultural dams have come into widespread existence since the 1940s (Renwick *et al* 2006), research into the role of these small artificial lentic systems in GHG cycles has only very recently materialized (Grinham *et al* 2018, Ollivier *et al* 2019, Webb *et al* 2019b). Here, 11 studies with $N_2O$ concentration measurements were collected. Water depth ranged from 0.18 to 5.1 m and land use included irrigated crops, mixed arable, rice, pastures, and palm oil plantations. Studies were distributed across China, Australia, UK, France, Canada, and Indonesia (figure S1). Overall, a mean EF of 0.13% (0.04%–0.46%) was found, which is half the IPCC $EF_{5r}$ estimate of 0.26% for which these systems are included. Instead, the agricultural pond EF found here is comparable to the EF for lakes, ponds, and reservoirs of 0.12% found in a previous review for the refinement of IPCC guidelines (Tian *et al* 2019). However, despite a significantly lower EF estimate compared to rivers (0.30%), lakes, ponds, and reservoirs were not assigned their own EF in the current IPCC $EF_5$ guidelines.

In terms of $N_2O$ concentrations, farm dams were significantly lower than all other artificial waters reviewed here (figure 2), reflecting findings found from a limited number of studies comparing different surface waters (Outram and Hiscock 2012, Xia *et al* 2013, Xiao *et al* 2019). Farm dams also had the only significant relationship with $NO_3$ out of the artificial waters (figure S2), where $N_2O$ increased on a logarithmic scale with increasing $NO_3$, suggesting a closer alignment to steady state N processing conditions. These differences may be because farm dams are less hydrologically dynamic compared with drainage water bodies due to their role in water storage. Higher water residence times allow for more time for runoff N to be transformed into $N_2O$, $NO_x$ or $N_2$ and can even be favourable for $N_2O$ consumption processes (Webb *et al* 2019a). For example, the development of strong thermoclines under steady hydrological conditions has been shown to be a strong influencing factor in supporting dissolved $N_2O$ undersaturation (Webb *et al* 2019a). Farm dams also have a lower gas transfer velocity compared to channels (Premaratne *et al* 2017, Ollivier *et al* 2019, Webb *et al* 2019a), leading to slower gas exchange between the water–atmosphere surface and potentially less

discrepancy between changes in $N_2O$ and $NO_3$ concentrations.

Introducing on-farm dams into intensive agricultural landscapes, especially those with a high density of tile drainage, may provide a measure to reduce potential $N_2O$ emissions further downstream. The strong reductive conditions enhanced due to low water velocity enables ponds to remove greater amounts of N than streams (Li *et al* 2013, Garnier *et al* 2014). This supports their ability to receive high inorganic N loads without the consequence of producing proportional $N_2O$ emissions (Webb *et al* 2019a). The high density of streams and ponds scattered within the rice-paddy-dominated watersheds, characteristic of China, is an example of where high N input does not always lead to high N concentrations in the surface water (Xia *et al* 2013, Xiong *et al* 2015). Further, a study of the Orgeval watershed in France explored the role of pond implementation in drainage areas and predicted a 34%–47% reduction in surface water N export (Passy *et al* 2012).

### 4.5. Low EF in waters with high N loading

Regression analysis between mean $N_2O$ concentrations and $NO_3$ concentrations revealed a significant positive-logarithmic rather than a linear response, and that in most cases $EF_{5g}$ and $EF_{5r}$ models overestimated $N_2O$ concentrations for artificial waters (figure 3(A)). A stronger relationship between $NO_3$ and EF was found where EF scales inversely with higher $NO_3$ (figure 3(B)). This relationship was consistent across subsurface drains, surface drains, and farm dams (figure S3), and suggests that $N_2O$ emissions in most artificial waters are overestimated using standard $EF_5$ modelling. Non-linearity in the $N_2O$ concentration to $NO_3$ response has been observed in rivers, implying a limit on $NO_3$ processing and subsequent $N_2O$ production (Turner *et al* 2016, Wang *et al* 2018). Moving away from stationary EFs and developing a model that accounts for the inverse scaling of EF with higher N loading will greatly reduce $N_2O$ emission uncertainties from highly N polluted agricultural waters.

This pattern of lower $N_2O$ production with increasing $NO_3$ loading is consistent with data from previous studies, although the processes driving this trend remains unclear. In agricultural watersheds receiving high N loads, less $N_2O$ may be produced relative to $NO_3$ due to biological saturation (Mulholland *et al* 2008, Beaulieu *et al* 2011, Xiao *et al* 2019). Alternatively, undersaturation of dissolved $N_2O$ in the presence of high $NO_3$ can occur if the right reductive conditions exist to facilitate complete denitrification, as seen in farm dams in Canada (Webb *et al* 2019a). Furthermore, in open water systems any $N_2O$ produced in excess of atmospheric equilibrium will degas into the atmosphere more rapidly than microbial $NO_3$ transformations. If much of the $N_2O$ has already degassed, then discrepancies in the $EF_5$ method that derives ratios between actual measured fluxes of $N_2O$ and $NO_3$ versus dissolved concentrations will arise (Clough *et al* 2006).

### 4.6. Artificial and natural waters in emission factor groups

To determine if artificial waters deserve recognition separate to natural waters for indirect $N_2O$ emission accounting, we compared $N_2O$ concentrations, $NO_3$, and EFs from artificial waters collected in this study with natural waters contributing to the current $EF_{5g}$ and $EF_{5r}$ models. In this analysis, irrigation canals were included with the surface drainage group, renamed to surface channels, due to the small number of studies and the fact that no significance difference was detected in previous analyses (figure 2). Natural waters were divided into groundwater, lakes, rivers, and streams as defined in Tian *et al* (2019). Despite finding significant differences in $N_2O$ concentrations, findings revealed no significant differences in EFs between artificial and natural waters (figure 4). This suggests that delineation of water groups into artificial and natural waters is unlikely to improve EF uncertainty within the current IPCC framework for assigning default EFs.

Mean $N_2O$ concentrations in farm dams were not significantly different between lakes, rivers, and streams, yet surface channels were significantly different to groundwater. Notable differences governing $N_2O$ concentrations in response to N loadings from artificial systems compared to natural waters have been found within agricultural watersheds. This included river $N_2O$ dynamics being more affected by heavy rainfall events than rice paddy ponds (Xiao *et al* 2019), and differences in redox conditions and wind turbulence affecting $N_2O$ production between ditches and a river (Outram and Hiscock 2012). In contrast, both $N_2O$ and $NO_3$ concentrations for subsurface drainage were comparable to groundwater (figure 4), supporting the assumption that leached soil pore water is the primary source of supersaturated $N_2O$ in both waters (Well *et al* 2005). On this basis, the IPCC EF grouping for $EF_{5g}$ is likely correct to include groundwater and subsurface drainage together, but not open water surface drainage (figure 4).

### 4.7. Limitations and future research

A number of competing factors that alter $N_2O$ and $NO_3$ concentrations, either separately or collectively, complicates the assumed linear response when estimating $N_2O$ emissions using default EFs. Some of these factors are more apparent in different artificial waters, such as higher gaseous losses in surface channels and where there is a greater capacity for $N_2O$ consumption, such as farm dams. As others have proven, a process-based model instead of revising default EFs will likely lead to the most noticeable improvements in estimating indirect aquatic $N_2O$ emissions

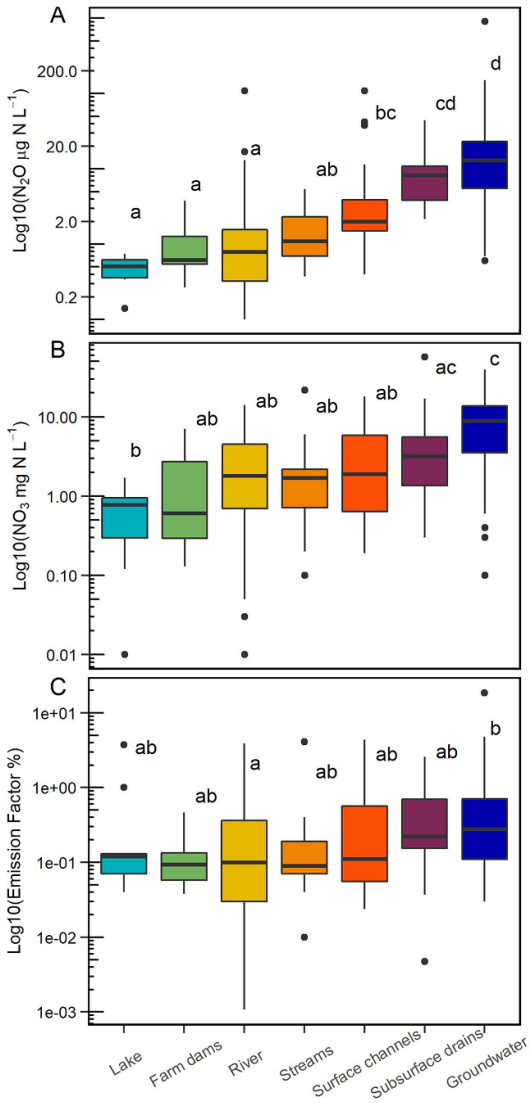


**Figure 4.** Variation of $\log_{10}$-transformed dissolved $N_2O$ concentration ($\mu$g N $l^{-1}$), $NO_3$ (mg N $l^{-1}$) concentrations, and EFs (%) between artificial and natural agricultural surface waters. Note irrigation canals have been combined with surface drains due to no significant differences observed in previous analyses (figure 2). Different letters represent significant differences ($p < 0.05$) among groups (one-way ANOVA and Tukey–Kramer post hoc test). Boxes span the interquartile range (25%–75% quartiles), whiskers 5%–95% of observations, horizontal line the median, and circle points represent outliers.

(Maavara *et al* 2019). However, the ability to develop process-based models for specific water types is limited by a severe lack of studies on artificial waters. Until then, a number of limitations with the current model of predicting $N_2O$ emissions needs to be considered.

Firstly, lack of consistent reporting of other important environmental factors that influence $N_2O$ production impedes development towards a process-based model. For example, proximity of open channels to agricultural fields may play a role in driving EFs, as stream order and distance from the N source have been shown to significantly impact $N_2O$ concentrations (Reay *et al* 2004a; Turner *et al* 2015). Many of the environmental variables have been found to drive $N_2O$ concentrations in artificial waters, including the DOC:$NO_3$ ratio, pH, fertilizer application, and precipitation, were tested across a subset of the artificial waters collected in this review (figure S4). The $N_2O$ concentration was inversely related to both DOC:$NO_3$ and pH, while annual precipitation was less strong as a predictor and no relationship was found with total N application. Variables such as DOC and pH are often found to be strong drivers of aquatic $N_2O$ dynamics, as at higher values they can represent conditions more favourable to microbial N reduction (Peacock *et al* 2017, Audet *et al* 2020). However, values for these factors could only be derived from 30% to 40% of studies reviewed here.

The large range in EFs within groups contributes to major uncertainty when applying default values to local sites. Grouping artificial waters into their respective groups also revealed a large range in EF, as well as a highly skewed distribution in $N_2O$, $NO_3$, and EF data (figure S5). Bias can arise when upscaling emissions using the mean statistic when the mean and median of a dataset are distinctly different, such as in surface drains with a mean and median EF of 0.46% and 0.11%, respectively (table S1). As discussed by others, care must be taken when extrapolating means derived from a skewed dataset to estimate GHG emissions (Grinham *et al* 2018; Rosentreter and Williamson 2020). Due to the low number of observations and geographical bias of studies collected here, the $N_2O$, $NO_3$, and EF datasets are unlikely to represent the 'real-world' distribution. In fact, ~50% of studies were located in the UK, which represents a notable limitation and a potentially sizeable source of error when extrapolating EFs globally. The choice of the arithmetic mean statistic in these cases may not provide a representative estimate for local-scale $N_2O$ emissions and may explain why the IPCC default EFs are often found to be overestimated.

Default EFs that rely on the mass ratio of $N_2O$ to $NO_3$ concentrations are also complicated by kinetic limitations introduced by variable gas exchange rates and water residence times across different water types. Open drains in particular represent complex aquatic systems for constraining $N_2O$ emissions due to their dynamic hydrology and variable design. Where rapid gaseous $N_2O$ losses occur within a water body, the $N_2O$:$NO_3$ ratio will be smaller as the biological processes consuming $NO_3$ operate at a slower time scale than physical evasion to the atmosphere. Additionally, water residence time introduces a hydrological constraint on the extent of in-system N transformations (Maavara *et al* 2019). Together, the physical constraints from gas transfer and hydrologic exchange result in a variable $N_2O$–N/$NO_3$–N concentration ratio that alters biological processing rates across different water types.

The IPCC provides two different methods for calculating indirect $N_2O$ EFs, as presented in the

introduction, and it is important to be aware of the different EF values these two methods may produce. There can be large discrepancy between EFs estimated from mass N-fluxes and EFs estimated from N-concentrations ratios (as reviewed here), as reported in Clough *et al* (2006) and Outram and Hiscock (2012), whereas at other times the two methods yield similar estimates (e.g. Hama-Aziz *et al* 2017, Premaratne *et al* 2017). In the few studies that reported $EF_5$ derived from both methods, or provided detailed information to allow the calculation, differences in the average EF ratio ranged from 0.00003 to 0.0469 in surface drainage systems and 0.0001 to 0.003 in subsurface drainage (supplementary table S3). Often, $N_2O$ fluxes were calculated using gas exchange models, which can be a large source of uncertainty if not produced specifically for the local water system. Development of models for calculating gas transfer velocity across specific artificial water types will advance the field by enabling standardised emissions upscaling of dissolved $N_2O$ concentrations. Some progress has been made by recent studies in this area, which report gas transfer coefficients of 3.8–6.6 m $d^{-1}$ for surface drains (Premaratne *et al* 2017) and 1–1.64 m $d^{-1}$ for on-farm dams (Ollivier *et al* 2019, Webb *et al* 2019).

A hybrid modelling approach that integrates the biological and chemical controls of $N_2O$ process rates with hydromorphological properties (e.g. Marzadri *et al* 2020) of artificial waters, including their type, area, water residence time, and distance from farm fields or runoff source, is likely required to capture the complexity of these systems. However, until these local-scale processes for artificial waters can be upscaled in modelling efforts, scaling EFs with $NO_3$ loading may prove to be a promising first step to move away from the default EF approach. Significant inverse correlations were found across all water groups (excluding irrigation canals) between EF and $NO_3$, while only farm dams revealed $NO_3$ to be a significant driver of $N_2O$ (figure S3). To develop this concept further, future research should explore $N_2O$ production thresholds within high $NO_3$ environments to better understand the conditions that lead to the inverse EF to $NO_3$ relationship. Some studies have started progressing this area, by investigating denitrification rates/changes in ditches receiving greater $NO_3$ inputs (Veraart *et al* 2017, She *et al* 2018), and looking at stratification in farm dams supporting $N_2O$ consumption under high $NO_3$ conditions (Webb *et al* 2019a). Another variable worth investigating is the $DOC/NO_3$ ratio, which we also found to be a significant inverse driver of $N_2O$ in artificial waters, as $DOC/NO_3$ can be a proxy for N limitation within aquatic ecosystems (Peacock *et al* 2017).

Finally, it is difficult to assess the relative impact of artificial waters on regional and global indirect $N_2O$ emissions until a database exists on the area and distribution these systems cover. The recent revised quantification of global $N_2O$ sources and sinks attributes 0.5 Tg N $yr^{-1}$ to indirect anthropogenic emissions from streams, rivers, lakes, reservoirs (>0.1 $km^2$), and estuaries, which is equivalent to 13% of total direct agricultural $N_2O$ emissions. However, this estimate does not include small artificial waters. This gap presents an opportunity to refine the contribution of anthropogenic $N_2O$ emissions in total global land emissions, where an estimated discrepancy of ~1.8 Tg N $yr^{-1}$ exists between bottom-up and top-down land models (Tian *et al* 2020). Explicitly including artificial waters in models for budget quantification will likely make a significant contribution to the agricultural $N_2O$ budget in regions where artificial waterbodies have been densely created, such as the irrigation networks in the Mediterranean region of Spain and irrigation ponds in southern China (Aguilera *et al* 2019, Chen *et al* 2019). Development of national inventories are already underway in some regions (Aguilera *et al* 2019, Malerba *et al* 2021), yet a collective country-by-country effort is needed to quantify the global extent.

## 5. Conclusion

Our analysis of the available literature suggests that subsurface drains, surface drains, and on-farm dams function differently in terms of surface water $N_2O$ production, while more studies are urgently needed to assess the role of irrigation canals. Studies showed that EFs vary largely in space and time within surface waters, which hinders detection of differences between artificial waters. The huge variability of EFs within water groups challenges the concept that $N_2O$ production in downstream waters is proportional to agricultural $NO_3$ load. The current IPCC methodology using generalised EFs overestimates indirect $N_2O$ production in waters receiving high $NO_3$ loading; a characteristic of many artificial waters. In fact, in these systems EFs scale downward with increasing N load. Improving the prediction of indirect $N_2O$ emissions from agricultural surface waters likely requires a move away from stationary EFs and explicit inclusion of different artificial waters. On a global scale, the lack of accounting for artificial waters in the most recent global $N_2O$ budget presents a significant gap in the quantification of indirect agricultural $N_2O$ emissions from aquatic systems (Tian *et al* 2020). As such, we suggest a need for a global database on the size, density, and distribution of artificial waters in agricultural landscapes, as well as a need to expand the global artificial waters $N_2O$ dataset beyond the UK. Further, future field measurements should target different artificial waters across major agricultural industries to advance the biogeochemical understanding to the level that has been achieved for rivers and lakes (Lauerwald *et al* 2019, Maavara *et al* 2019).

## Data availability statement

All data that support the findings of this study are included within the article (and any supplementary files).

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## References

Aguilera E, Vila-Traver J, Deemer B R, Infante-Amate J, Guzmán G I and González De Molina M 2019 Methane emissions from artificial waterbodies dominate the carbon footprint of irrigation: a study of transitions in the food–energy–water–climate nexus (Spain, 1900–2014) *Environ. Sci. Technol.* **53** 5091–101

Audet J *et al* 2020 Forest streams are important sources for nitrous oxide emissions *Glob. Change Biol.* **26** 629–41

Beaulieu J J *et al* 2011 Nitrous oxide emission from denitrification in stream and river networks *Proc. Natl Acad. Sci.* **108** 214–9

Billen G, Garnier J, Grossel A, Thieu V, Théry S and Hénault C 2020 Modeling indirect $N_2O$ emissions along the N cascade from cropland soils to rivers *Biogeochemistry* **148** 207–21

Blann K L, Anderson J L, Sands G R and Vondracek B 2009 Effects of agricultural drainage on aquatic ecosystems: a review *Crit. Rev. Environ. Sci. Technol.* **39** 909–1001

Castellano M J, Archontoulis S V, Helmers M J, Poffenbarger H J and Six J 2019 Sustainable intensification of agricultural drainage *Nat. Sustain.* **2** 914–21

Chen W, He B, Nover D, Lu H, Liu J, Sun W and Chen W 2019 Farm ponds in southern China: challenges and solutions for conserving a neglected wetland ecosystem *Sci. Total Environ.* **659** 1322–34

Christen E W, Ayars J E and Hornbuckle J W 2001 Subsurface drainage design and management in irrigated areas of Australia *Irrig. Sci.* **21** 35–43

Chumchal M M, Drenner R W and Adams K J 2016 Abundance and size distribution of permanent and temporary farm ponds in the southeastern Great Plains *Inland Waters* **6** 258–64

Clagnan E, Thornton S F, Rolfe S A, Tuohy P, Peyton D, Wells N S and Fenton O 2018 Influence of artificial drainage system design on the nitrogen attenuation potential of gley soils: evidence from hydrochemical and isotope studies under field-scale conditions *J. Environ. Manage.* **206** 1028–38

Clifford C and Heffernan J 2018 Artificial aquatic ecosystems *Water* **10** 1096

Clough T J, Bertram J E, Sherlock R R, Leonard R L and Nowicki B L 2006 Comparison of measured and $EF_5$-r-derived $N_2O$ fluxes from a spring-fed river *Glob. Change Biol.* **12** 477–88

Dinnes D L, Karlen D L, Jaynes D B, Kaspar T C, Hatfield J L, Colvin T S and Cambardella C A 2002 Nitrogen management strategies to reduce nitrate leaching in tile-drained midwestern soils *Agron. J.* **94** 153–71

Dowdell R, Burford J and Crees R 1979 Losses of nitrous oxide dissolved in drainage water from agricultural land *Nature* **278** 342–3

Fu C, Lee X, Griffis T J, Baker J M and Turner P A 2018 A modeling study of direct and indirect $N_2O$ emissions from a representative catchment in the U.S. corn belt *Water Resour. Res.* **54** 3632–53

Garnier J, Billen G, Vilain G, Benoit M, Passy P, Tallec G, Tournebize J, Anglade J, Billy C and Mercier B 2014 Curative vs. preventive management of nitrogen transfers in rural areas: lessons from the case of the Orgeval watershed (Seine River basin, France) *J. Environ. Manage.* **144** 125–34

Grinham A, Albert S, Deering N, Dunbabin M, Bastviken D, Sherman B, Lovelock C E and Evans C D 2018 The importance of small artificial water bodies as sources of methane emissions in Queensland, Australia *Hydrol. Earth Syst. Sci.* **22** 5281–98

Hama-Aziz Z Q, Hiscock K M and Cooper R J 2017 Dissolved nitrous oxide ($N_2O$) dynamics in agricultural field drains and headwater streams in an intensive arable catchment *Hydrol. Process.* **31** 1371–81

Harrison J A, Matson P A and Fendorf S E 2005 Effects of a diel oxygen cycle on nitrogen transformations and greenhouse gas emissions in a eutrophied subtropical stream *Aquat. Sci.* **67** 308–15

Hasegawa K, Hanaki K, Matsuo T and Hidaka S 2000 Nitrous oxide from the agricultural water system contaminated with high nitrogen *Chemosphere* **2** 335–45

Hénault C, Grossel A, Mary B, Roussel M and Léonard J 2012 Nitrous oxide emission by agricultural soils: a review of spatial and temporal variability for mitigation *Pedosphere* **22** 426–33

Hergoualc'h K, Akiyama H, Bernoux M, Chirinda N, Del Prado A, Kasimir Å, MacDonald J D, Ogle S M, Regina K and Van Der Weerden T J 2019 $N_2O$ emissions from managed soils, and $CO_2$ emissions from lime and urea application *Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories*

Herzon I and Helenius J 2008 Agricultural drainage ditches, their biological importance and functioning *Biol. Conserv.* **141** 1171–83

Jamali H, Quayle W, Scheer C, Rowlings D and Baldock J 2016 Effect of soil texture and wheat plants on $N_2O$ fluxes: a lysimeter study *Agric. For. Meteorol.* **223** 17–29

Kitamura K and Nakaya T 2010 Case study on application of asset management of irrigation infrastructure for rice production in Australia *Japan Agric. Res. Q.* **44** 45–51

Lauerwald R, Regnier P, Figueiredo V, Enrich-Prast A, Bastviken D, Lehner B, Maavara T and Raymond P 2019 Natural lakes are a minor global source of $N_2O$ to the atmosphere *Glob. Biogeochem. Cycles* **33** 1564–81

Lecce S A, Pease P P, Gares P A and Wang J 2006 Seasonal controls on sediment delivery in a small coastal plain watershed, North Carolina, USA *Geomorphology* **73** 246–60

Li X, Xia Y, Li Y, Kana T M, Kimura S D, Saito M and Yan X 2013 Sediment denitrification in waterways in a rice-paddy-dominated watershed in eastern China *J. Soils Sediments* **13** 783–92

Maavara T, Lauerwald R, Laruelle G G, Akbarzadeh Z, Bouskill N J, Van Cappellen P and Regnier P 2019 Nitrous oxide emissions from inland waters: are IPCC estimates too high? *Glob. Change Biol.* **25** 473–88

Macdonald B C T, Nadelko A, Chang Y, Glover M and Warneke S 2016 Contribution of the cotton irrigation network to farm nitrous oxide emissions *Soil Res.* **54** 651–8

Malerba M E, Wright N and Macreadie P I 2021 A continental-scale assessment of density, size, distribution and historical trends of farm dams using deep

learning convolutional neural networks *Remote Sens.* **13** 319
Marzadri A, Tonina D and Bellin A 2020 Power law scaling model predicts $N_2O$ emissions along the upper Mississippi River basin *Sci. Total Environ.* **732** 138390
Meyer D, Dimitriadou E, Hornik K, Weingessel A and Leisch F 2019 e1071: misc functions of the department of statistics, probability theory group (formerly: E1071), TU Wien. R package version 1.7–3 (available at: https://CRAN.R-project.org/package=e1071)
Molden D 2007 *Water for Food Water for Life: A Comprehensive Assessment of Water Management in Agriculture* (London: Routledge)
Mulholland P J, Helton A M, Poole G C, Hall R O, Hamilton S K, Peterson B J, Tank J L, Ashkenas L R, Cooper L W and Dahm C N 2008 Stream denitrification across biomes and its response to anthropogenic nitrate loading *Nature* **452** 202–5
Neumann K, Stehfest E, Verburg P H, Siebert S, Müller C and Veldkamp T 2011 Exploring global irrigation patterns: a multilevel modelling approach *Agric. Syst.* **104** 703–13
Ollivier Q R, Maher D T, Pitfield C and Macreadie P I 2019 Punching above their weight: large release of greenhouse gases from small agricultural dams *Glob. Change Biol.* **25** 721–32
Outram F N and Hiscock K M 2012 Indirect nitrous oxide emissions from surface water bodies in a lowland arable catchment: a significant contribution to agricultural greenhouse gas budgets? *Environ. Sci. Technol.* **46** 8156–63
Passy P, Garnier J, Billen G, Fesneau C and Tournebize J 2012 Restoration of ponds in rural landscapes: modelling the effect on nitrate contamination of surface water (the Seine River Basin, France) *Sci. Total Environ.* **430** 280–90
Peacock M, Ridley L M, Evans C D and Gauci V 2017 Management effects on greenhouse gas dynamics in fen ditches *Sci. Total Environ.* **578** 601–12
Perego A, Basile A, Bonfante A, De Mascellis R, Terribile F, Brenna S and Acutis M 2012 Nitrate leaching under maize cropping systems in Po Valley (Italy) *Agric. Ecosyst. Environ.* **147** 57–65
Premaratne M, Clough T J and Kelliher F M 2017 Determining the nitrous oxide transfer velocity and emission factor of an agricultural drain *N. Z. J. Agric. Res.* **60** 277–86
R Core Team 2020 *R: A Language and Environment for Statistical Computing* (Vienna: R Foundation for Statistical Computing) (available at: www.R-project.org/)
Reay D S, Edwards A C and Smith K A 2004a Determinants of nitrous oxide emission from agricultural drainage waters *Water Air Soil Pollut.* **4** 107–15
Reay D S, Smith K A and Edwards A C 2003 Nitrous oxide emission from agricultural drainage waters *Glob. Change Biol.* **9** 195–203
Reay D S, Smith K A and Edwards A C 2004b *Biogeochemical Investigations of Terrestrial, Freshwater, and Wetland Ecosystems across the Globe* (Berlin: Springer) pp 437–51
Renwick W H, Sleezer R O, Buddemeier R W and Smith S V 2006 Small artificial ponds in the United States: impacts on sedimentation and carbon budget *Proc. Eighth Federal Interagency Sedimentation Conf.* pp 738–44
Roper J D, Burton D L, Madani A and Stratton G W 2013 A simple method for quantifying dissolved nitrous oxide in tile drainage water *Can. J. Soil Sci.* **93** 59–64
Rosentreter J A and Williamson P 2020 Concerns and uncertainties relating to methane emissions synthesis for vegetated coastal ecosystems *Glob. Change Biol.* **26** 5351–2
Sawamoto T, Kusa K, Hu R and Hatano R 2002 Dissolved $N_2O$, $CH_4$, and $CO_2$ in pipe drainage, seepage, and stream water in a livestock farm in Hokkaido, Japan *Soil Sci. Plant Nutr.* **48** 433–9
Sawamoto T, Kusa K, Hu R and Hatano R 2003 Dissolved $N_2O$, $CH_4$, and $CO_2$ emissions from subsurface-drainage in a structured clay soil cultivated with onion in Central Hokkaido, Japan *Soil Sci. Plant Nutr.* **49** 31–38
Schultz B and De Wrachien D 2002 Irrigation and drainage systems research and development in the 21st century *Irrig. Drain.* **51** 311–27
Schwenke G, Nachimuthu G, Mercer C and McPherson A 2020 Tail-drain sediments are a potential hotspot for nitrous oxide emissions in furrow-irrigated Vertisols used to grow cotton: a laboratory incubation study *J. Environ. Qual.* **49** 14–26
Seitzinger S P, Kroeze C and Styles R V 2000 Global distribution of $N_2O$ emissions from aquatic systems: natural emissions and anthropogenic effects *Chemosphere* **2** 267–79
She D L, Chen X Y, Gao X M, Zhang W J and Xia Y Q 2018 Impact of exogenous nitrogen import on sediment denitrification and $N_2O$ emissions in ditches under different land uses *Huan Jing Ke Xue* **39** 3689–95
Siegfried F, Arne G, Arne J, Joakim P and Lars S 1994 Nitrogen removal in created ponds *Ambio* **23** 349–57
Soana E, Bartoli M, Milardi M, Fano E A and Castaldelli G 2019 An ounce of prevention is worth a pound of cure: managing macrophytes for nitrate mitigation in irrigated agricultural watersheds *Sci. Total Environ.* **647** 301–12
Thorburn P J, Biggs J S, Attard S J and Kemei J 2011 Environmental impacts of irrigated sugarcane production: nitrogen lost through runoff and leaching *Agric. Ecosyst. Environ.* **144** 1–12
Tian H *et al* 2020 A comprehensive quantification of global nitrous oxide sources and sinks *Nature* **586** 248–56
Tian L, Cai Y and Akiyama H 2019 A review of indirect $N_2O$ emission factors from agricultural nitrogen leaching and runoff to update of the default IPCC values *Environ. Pollut.* **245** 300–6
Turner P A, Griffis T J, Lee X, Baker J M, Venterea R T and Wood J D 2015 Indirect nitrous oxide emissions from streams within the US corn belt scale with stream order *Proc. Natl Acad. Sci.* **112** 9839–43
Turner P, Griffis T, Baker J, Lee X, Crawford J T, Loken L C and Venterea R 2016 Regional-scale controls on dissolved nitrous oxide in the upper Mississippi River *Geophys. Res. Lett.* **43** 4400–7
Veraart A J, Dimitrov M R, Schrier-Uijl A P, Smidt H and De Klein J J M 2017 Abundance, activity and community structure of denitrifiers in drainage ditches in relation to sediment characteristics, vegetation and land-use *Ecosystems* **20** 928–43
Verstraeten G and Poesen J 2000 Estimating trap efficiency of small reservoirs and ponds: methods and implications for the assessment of sediment yield *Prog. Phys. Geogr. Earth Environ.* **24** 219–51
Waller P and Yitayew M 2015 *Irrigation and Drainage Engineering* (Berlin: Springer)
Wang M, Li Y F, Lei K and Yang L B 2018 Concentration, flux, and emission factor of $N_2O$ in rivers with different nitrogen pollution features *Huan Jing Ke Xue* **39** 5400–9
Wang X, He Y, Yuan X, Chen H, Peng C, Yue J, Zhang Q, Diao Y and Liu S 2017 Greenhouse gases concentrations and fluxes from subtropical small reservoirs in relation with watershed urbanization *Atmos. Environ.* **154** 225–35
Webb J R, Hayes N M, Simpson G L, Leavitt P R, Baulch H M and Finlay K 2019a Widespread nitrous oxide undersaturation in farm waterbodies creates an unexpected greenhouse gas sink *Proc. Natl Acad. Sci.* **116** 9814–9
Webb J R, Leavitt P R, Simpson G L, Baulch H M, Haig H A, Hodder K R and Finlay K 2019b Regulation of carbon dioxide and methane in small agricultural reservoirs: optimizing potential for greenhouse gas uptake *Biogeosciences* **16** 4211–27
Well R, Weymann D and Flessa H 2005 Recent research progress on the significance of aquatic systems for indirect agricultural $N_2O$ emissions *Environ. Sci.* **2** 143–51
White S A, Santos I R, Conrad S R, Sanders C J and Hessey S 2021 Large aquatic nitrous oxide emissions downstream of intensive horticulture driven by rain events *J. Hydrol.* 126066

Xia Y, Li Y, Ti C, Li X, Zhao Y and Yan X 2013 Is indirect $N_2O$ emission a significant contributor to the agricultural greenhouse gas budget? A case study of a rice paddy-dominated agricultural watershed in eastern China *Atmos. Environ.* **77** 943–50

Xiao Q, Hu Z, Fu C, Bian H, Lee X, Chen S and Shang D 2019 Surface nitrous oxide concentrations and fluxes from water bodies of the agricultural watershed in eastern China *Environ. Pollut.* **251** 185–92

Xiong Y, Peng S, Luo Y, Xu J and Yang S 2015 A paddy eco-ditch and wetland system to reduce non-point source pollution from rice-based production system while maintaining water use efficiency *Environ. Sci. Pollut. Res.* **22** 4406–17

Yao Y, Tian H, Shi H, Pan S, Xu R, Pan N and Canadell J G 2020 Increased global nitrous oxide emissions from streams and rivers in the Anthropocene *Nat. Clim. Change* **10** 138–42