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A Review of Greenhouse Gas Emissions from Selected Wetlands in the Vietnamese Mekong Delta

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Summary

Nearly half of Vietnam's wetlands are concentrated in the Vietnamese Mekong Delta. Wetlands play an important role in carbon storage while contributing to Vietnam's biological diversity and economy. This report reviews the wider literature to understand how land use within the Vietnamese Mekong Delta and likely management scenarios can impact greenhouse gas emissions. The report is structured into six main sections covering an introduction, key characteristics of the Vietnamese Mekong Delta wetlands, literature-based greenhouse gas emissions assessments, a review of commonly available models for estimating greenhouse gas emissions, estimations of emissions using a select number of reviewed models, with a focus on paddy rice, and finally ends with some conclusions.

Contents

1. Introduction

Wetlands cover 6% of the global land surface, and they account for 12% of the global carbon pool (Erwin 2008). Various definitions of wetlands are provided in the literature. RAMSAR (2007) defines it as "areas where water is the primary factor controlling the environment and the associated plant and animal life. They occur where the water table is at or near the surface of the land, or where the land is covered by water." Another definition provided by Wetlands International (Nd) states that they "occur wherever water meets land. These unique habitats include mangroves, peatlands and marshes, rivers and lakes, deltas, floodplains and flooded forests, rice-fields, and even coral reefs." Total wetlands in Vietnam encompass nearly 10 M ha (Thanh and Yabar 2015). The Vietnamese Mekong Delta (VMD) is a significant wetland ecosystem that contributes to Vietnam's biological diversity and economy. There is a strong co-dependence across the delta- from the flush of sediment that moves through the system, the tidal flows, and the interconnectedness of these. Modern modifications to water movement within the system risk its functionality, and very little research has been done to understand the long-term impacts of these. Water resources in the area and the livelihoods dependent on the system are at risk due to the anticipation of climate change. Unique habitats such as mangroves, which provide valuable ecosystem services, are threatened by land use changes. However, the national food security and the national budget of the VMD benefit from the income of large industries based around fisheries and aquaculture. The present review here seeks to collate the literature around land use within the VMD and likely management scenarios, allowing us to explore the greenhouse gas (GHG) emissions potential of land use activities. The key objective of this review is to inform discussion around climate mitigation opportunities, as well as the long-term sustainability of the system.

This report consists of five main sections after this introduction: Section two - Wetland characteristics in the VMD; section three - emissions from wetlands in the Mekong Delta; section four - review of models for estimating GHG emissions; section five - GHG emissions from potential land use change in the Mekong Delta, and section six - conclusions.

2. Wetland characteristics in the Vietnamese Mekong Delta

2.1 Land Use Classes Across the VMD

The Vietnamese Mekong Delta (VMD) wetland area covers nearly 5 M ha of land (Table 1). It consists of an ecosystem abundant with tidal floodplains, peatland marsh, coastal marsh, and estuaries within the territory. These wetlands could be split into different types based on their characteristics. Typically, the majority of VMD wetlands are freshwater wetlands, and their area is nearly 40% of the total wetlands in VMD. Saline lagoons are found least in VMD, and it is only 2521 ha as an area (Tung and Dap 2020). The following table (Table 1) illustrates the areas of typical wetland types in VMD.

Table 1. Areas of typical wetland types of the Vietnamese Mekong Delta

Wetland Type	Area (ha)	Percentage from total wetland area (%)
Freshwater wetlands	1,963,240	40.20
Saline coastal wetlands	1,636,069	33.50
Saline estuarine wetlands	1,052,102	21.54
Marshes and Swamps	229,363	4.70
Saline lagoons	2,521	0.05

Source: Tung and Dap (2020)

Other studies have used different names for different wetland categories and reported different numbers, e.g., Table 2 (Torell et al. 2001). Some studies further break this classification down to specific uses, such as open water, mangrove, deciduous forest, evergreen forest, mixed forest, built-up, single rice crop, double rice crop, triple rice crop, crops other than rice, barren land, aquaculture, grassland, and wetland (Mondal et al. 2022). According to the Mekong Delta Plan, the VMD is generally divided into upstream delta flood plains and downstream delta (Tran et al. 2019). The upper area of the VMD is dominated by rice cultivation. In between this area and the coast lay areas of seasonally flooded grassland (Triet nd). The lower coastal region of the delta is characterized by aquaculture and intact mangrove systems and has been broadly categorized as dense mangroves, sparse mangroves, aquaculture farms, arable land with crop cover, arable land without crop cover, settlements, and water bodies (Hong et al. 2019). Seasonal flooding ensures significant sediment deposition into the delta, which is vital for continued successful production. Dikes, sluice gates and water supply canal infrastructure were installed to prevent flooding and inundation from salt water, thus maintaining the suitability of the area for cropping (Mondal et al. 2022). Therefore, the government planning over the regulation of freshwater hydrology is one of the critical factors that have led to the successful development of the area (Mondal et al. 2022).

Table 2. Area coverage of wetland types of the Vietnamese Mekong Delta

Wetland type	Area (ha)
Floodplain wet rice	2,123,330
Marine sub-tidal	1,040,660
Seasonally flooded grassland	400,260
Perennial river	134,420
Intertidal estuarine aquaculture	126,220
Estuarine mangrove swamp	123,670
Seasonal freshwater swamp trees	122,790
Other land uses	< 20,000

Source: Torell et al. (2001)

2.2 Land Use Practices

Vietnam's fishery and aquaculture industry accounts for approximately 5% of the national Gross Domestic Product (GIZ 2021). Fisheries within the VMD are dominated by both subsistence and small-scale fisheries, with fishermen in the rivers and reservoirs or rice farmers making use of their fields and small canals for fishing (Torell et al. 2001), as well as large-scale industrial fishing and processing of products. Marine fisheries of the VMD are located within both the river estuaries and the South China Sea (Tuan et al. 1998). Alongside fisheries, other aquatic-based food is derived within the VMD- such as frogs, snails, crabs, shrimps, and insects. Aquaculture, specifically shrimp farming, has shown significant growth over the past decades (over 50% from 2010-2017), often integrated with agriculture systems in some areas within VMD (Dang 2020).

The cultivation of rice in the delta plays a key role in the country's (and region's) food security (Torell et al. 2001). Rice can be grown in up to three seasons in the year- Winter-Spring, Summer-Autumn, and Autumn-Winter (Clauss et al. 2018). Within the delta, the dike management (and farmer preference) tends to figure out whether farmers grow rice either once, twice, or thrice in the year. Double rice cropping and aquaculture are the top two land use categories in the VMD (Mondal et al. 2022), although areas of triple rice have been shown to increase significantly in recent years as it replaces double rice (Vu et al. 2022). However, there are sustainability issues around the expansion of triple rice, especially in relation to climate, water, and land (Mondal et al. 2022). The construction of dikes to sustain triple rice cropping in the upper areas of the VMD appears to have a significant impact on the annual flooding regime, which could lead to disastrous effects on the broader use of the delta (Vu et al. 2022). Temperature and precipitation particularly limit the viability of triple rice going forward, as there has been a 31% reduction in rainfall in the triple rice crop area (over 2000-2018) in A Giang and Dong Thap provinces, aligned with the general precipitation decline across the delta (27% over the same time period) (Mondal et al. 2022). Some farmers have already been reported to have changed their cropping practices in 2015-2018 away from rice entirely- potentially due to drought, which affected rice productivity (Mondal et al. 2022).

2.3 Changes in Land Use

Drivers of change in land management across the VMD can be attributed to increased exportation of food products produced within the delta, population growth and urbanization, intensification of agriculture (increased fertilizer and pesticide use), changes in sea level and increased salinization (Drogoul et al. 2016).

Installing high dikes encourages the intensification of rice production, preventing flood waters from entering fields during flood season. While this allows farmers to access a third yield and thus improve their income, there are noticeable determinantal effects from this intensification and change in hydrological management. The installation of high dikes has led to reductions in water retention capacity, increased flood risk, diminished dry season flows (exacerbating saltwater intrusion), disrupted ecosystem service flows, and prevented refreshing of fertile sediments and wild fish into rice fields (Tran et al. 2018). In addition, the productivity of triple rice farming systems has been shown to diminish over time, as farmer profits have been reported to reduce over a 15-year period (Tran et al. 2018).

The costs of production, 58-91% higher in triple rice compared to in the lower dike/double rice crop areas due to the higher level of fertilizer and pesticide inputs required, increase over time as the system becomes less productive. While the profitability of the triple rice system was shown to be 57% higher initially, this was reduced to just 6% higher after 15 years (Tran et al. 2018). Tran et al. 2018 also examined the benefits of alternative farming and found that mixed farming systems (e.g., rice and vegetables or fisheries) could increase farmer income by 12-268% whilst maintaining the environmental integrity of the system.

Along the coastal areas dominated by aquaculture, expansion of shrimp farming causes increased loss and degradation of mangrove systems (Hong et al. 2019). Other drivers of mangrove loss in this area are the collection of wood for fuel, cutting trees for house construction materials and rapid urbanization. The continued mismanagement of these systems has hampered the government's efforts to encourage sustainable mangrove use.

2.4 Summary of Wetland Characteristics

We can broadly classify critical ecosystems and land management within the delta based on the dominant land use and future scenarios (Table 3). From these, seasonally flooded grassland and mangrove areas are currently under the threat of conversion to rice and shrimp aquaculture, respectively.

Within each of these wetland types, a level of heterogeneity can be expected, e.g., due to different species, soils, and history of land management. Significant variability in the level of disturbance, recovery from disturbance (both natural and plantations), species and coastal versus inland environmental influences within mangrove areas can be identified from the remotely sensed data of mangrove systems (Vo and Kuenzer 2012), although this will likely still require some element of ground-truthing.

Table 3. Proposed classification of key land uses within the Vietnamese Mekong Delta

Area of the delta	Wetland type	Wetland use	Vegetation cover	Soil type
Upper	Floodplain wet rice	Triple rice	Rice	Alluvial soils
	Floodplain wet rice	Double rice	Rice	Alluvial soils
	Floodplain wet rice	Rice and vegetable crop	Rice and vegetables	Alluvial soils
Lower	Seasonally flooded grassland	Grassland	Grassland	Acid sulphate soil
	Mangrove	Aquaculture	None	Saline soils
	Mangrove	Degraded mangrove	Degraded mangrove forest	Saline soils
	Mangrove	Intact mangrove vegetation	Mangrove forest	Saline soils

Source: Miller et al. (1999)

3. Emissions from wetlands in the Mekong Delta

The greenhouse gas emissions of the wetland types described in Table 2 can be explored in this section.

3.1 Emissions from Rice Cultivation

3.1.1 Background

Driven by national policy (Linh and Thang 2015), there has been extensive, long-term expansion of rice production within the VMD, facilitated through hydrological management of the area with dams, canals, sluice gates, saline intrusion flood gates and monitoring of water quality, sedimentation, and soils (White 2002). Recent policies, such as the "1 Must Do and 5 Reductions' have also influenced rice management and expansion in an attempt to adjust unsustainable rice production practices such as poor water management and overuse of pesticides and fertilizers (Flor et al. 2021).Under its Nationally Determined Contributions to the United Nations Framework Convention on Climate Change, Vietnam states its efforts to reduce GHG emissions and include the withdrawal of water in the middle of cropping cycles to reduce methane (CH_4) emissions, as well as improved conversion of rice land (Socialist Republic of Vietnam 2022). Measures to reduce CH_4 emissions in sub-sectors of agriculture, especially wet rice farming, is a key focus for implementing Vietnam's stated efforts to reduce CH_4 emissions by 30% of 2020 levels by 2030 over at least 1.2 M ha of land (Torbick et al. 2017; Socialist Republic of Vietnam 2022). Conversion to more diversified systems, especially higher value crops and allowing for the natural flood regime to be restored are also promoted, with examples of evidence-based suggestions for diversification including single rice with a four-season squash, double locus crop with tourism, floating rice with vegetables, or triple rice but with a fishpond or vegetables (Tran et al. 2018). Land use for rice cultivation classified into single, double, and triple rice crops in VMD mapped using Sentinel-1 and Sentinel-2 satellite data is represented in Figure 1.

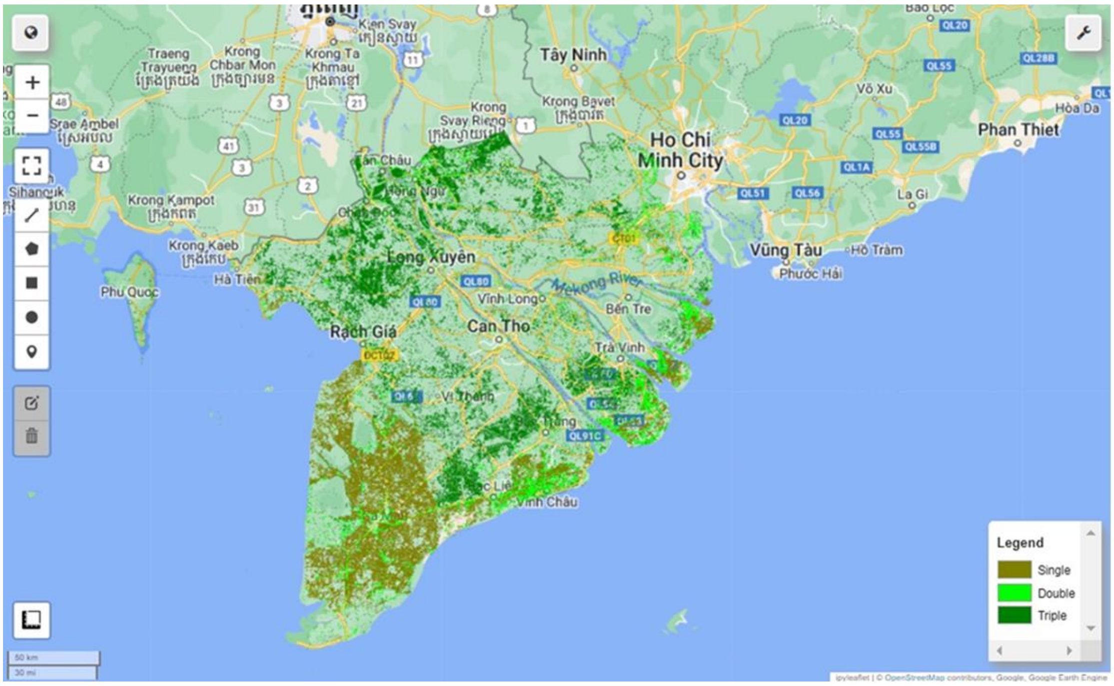

Figure 1. Rice Crop Map of Vietnamese Mekong Delta [*Source*: Ghosh et. al. (2022)]

3.1.2 Rice Management

Substantially higher rates of agrochemical use for rice production in triple cropping rice systems have been reported, with 30% of fertilizer use higher than in low-dike farming systems, equivalent to 100-200 kg ha^{-1} $crop^{-1}$, and up to 90% higher in some areas [300-500 kg ha^{-1} $crop^{-1}$ (Tran et al. 2018)]. The use of pesticides has also been reported to be higher in high dike systems, in one study rising 5% (4-12 bottles ha^{-1} $crop^{-1}$) to 39% (16-25 bottles ha^{-1} $crop^{-1}$) (Tran et al. 2018). The application of nitrogen (N) fertilizer in the form of N-N_2O (nitrous oxide) for continuously flooded rice results in an emission factor of 0.3%, as determined by the Intergovernmental Panel (Vo et al. 2020). Consequently, it can be inferred that the elevated usage of N fertilizer is unlikely to have a substantial impact on greenhouse gas (GHG) emissions. Low or negligible N_2O fluxes have been reported from multiple studies in the rice systems of the VMD (Drogoul et al. 2016; Vu et al. 2022)., including studies that captured seasonal variability in emissions (Oo et al. 2013) likely due to suppression of nitrification-denitrification processes due to anaerobic conditions.

3.1.3 Emissions from Rice

There are some key influencing features for consideration when exploring the GHG dynamics of rice in the Mekong Delta.

Seasonality

Season-based GHG emission factors have been found to be more significant than agro-zone based emission factors. This could be likely due to both land management decisions, such as the avoidance of adverse seasonal effects through adjusting cropping calendars, protection of rice-grown areas from adverse seasonal effects through improved infrastructure in canals and sluices (Vo et al. 2020), management of soils [such as incorporation of straw (Nauditt and Ribbe 2011)] as well as greater difference in seasonal environmental conditions such as temperature. Based on a study of 36 sites across the rice growing areas of Vietnam and covering 73 different cropping seasons, daily emission rates were found to be not significantly different between the edapho-hydrological zones of the Mekong Delta, with season-specific effects superseding zone-specific effects on CH_4 emissions (Vo et al. 2020). The same study identified that there were significant differences between emission factors in the seasons for both the Northern and Southern regions (although not the Central region) and recommended that different emission factors should be applied for different seasons in the North and South and not needed for the Central region (Vo et al. 2020). Several studies based on the seasonality of rice crops identify the significance of applying the emission factors rather than one emission factor for the whole region.

In a study in northern Vietnam across three crops in the year (rice-maize-rice), total cumulative CH_4 emissions were between 2 to 2.5 times higher during the summer rice season than during the spring rice season, with no emission peaks in the initial period after transplanting in the spring season (compared to the summer season which had a pronounced peak in emissions 9 days after transplanting the rice (Vu et al. 2015). This was assumed to be due to the low air temperature during the spring season. Contrasting seasonal patterns of CH_4 emission were also observed in another study in Northern Vietnam, this time in a double-cropped rice system where CH_4 flux peak occurred in the initial period after planting in the summer, but the peak was not as high in the spring rice crop and was observed in the middle to later growing periods (Oo et al. 2013). This was likely due to high soil temperature and, therefore, high soil organic matter turnover due to increased microbial activities, decomposition of recent decaying plant residues from shed leaves and root turnover, and high availability of root exudates in the rhizosphere (Oo et al. 2013).

CH_4 emissions are significantly lower in the spring season than in the summer season, and rice yields are higher (Vu et al. 2015). It has been identified that total cumulative CH_4 emissions were between 2 and 2.5 times higher in the summer rice season than in the spring rice season (Vu et al. 2015). This can give a much lower yield-scaled global warming potential (GWP) in the spring rice season than in the summer; the yield-scaled GWP for spring rice was approximately 1 kg CO_2 e lower per kg of rice grain than in the summer across multiple organic amendments treatments in a northern Vietnam study (Vu et al. 2015). A study in a double rice cropped system in the Red River Delta, Vietnam, also reported CH_4 emissions to be higher in the summer than in the spring but found higher N_2O fluxes observed in the spring than in the summer, which generally coincided with fertilization and drainage events (Tariq et al. 2017). Tirol-Padre et al. in Nauditt and Ribbe (2011) observed seasonal variability in CH_4 emissions, but also between the same seasons in consecutive years, which was attributed to farmer management through the incorporation of incompletely burnt rice straw residues prior to the summer season and then the removal of rice straw from the field the following year (Nauditt and Ribbe 2011). In addition to seasonality, other factors related to rice cultivation, like water table, soil type, etc., affect GHG emissions.

Water table

The water table is a key environmental feature that significantly affects CH_4 emissions due to the anaerobic digestion of organic material (Tariq et al. 2017). Hydrological management of rice paddies has the potential for significant CH_4 emissions reductions. A study (Tariq et al. 2017) comparing inefficient and efficient water management in rice paddies demonstrated higher GWP (through CH_4 and N_2O emissions) in inefficient systems compared to efficient ones. Continuous flooding in the inefficient water management scenario resulted in the highest GWP of all other scenarios (mid-season drainage, pre-planting plus mid-season drainage and early season plus mid-season drainage). A noticeable dynamic modification of conventional water practices is essential for mitigating CH_4 emissions, for example, managing water levels after incorporating rice residues, which can generate higher CH_4 emissions if allowed to decompose anaerobically (Tariq et al. 2017). Other water management techniques such as Alternative Wetting and Drying have been shown to reduce CH_4 emission significantly across all seasons, with one study demonstrating an average 71% reduction in emissions compared to those from continuous flooding (with no significant effect on N_2O emissions, Nauditt and Ribbe 2011).

Soil conditions

Soil conditions and environmental variables can also significantly affect CH_4 emissions. Oo et al. 2013 demonstrated a significant correlation between CH_4 emissions and soil temperature in a study in northwestern Vietnam. They also demonstrated significantly negative correlations between CH_4 emission and soil Eh in almost all treatments and a negative correlation between surface water pH and CH_4 emissions (Oo et al. 2013).

Studies using a model-based approach to estimate CH_4 emissions likely highlight other features of significance due to the model structure and key parameters. Researchers applied the Denitrification Decomposition (DNDC) model to the Red River Delta and found that the most significant drivers of rice methane emissions in continuously flooded areas as soil texture (clay fraction), soil organic carbon, and temperature and residue fraction to a lesser extent (Torbick et al. 2017).

Soil Amendments

Carbon inputs from organic amendments such as fresh manure, composted manure, digestate, biochar and composted digestate with rice straw, as well as N fertilizer, can all lead to increased CH_4 emissions due to the increased C input to the soil. Vu et al. reported this influence in their 2015 study, although the specific amendments based on a combination of liquid digestate, biochar and nitrogen did not lead to as significant CH_4 emissions compared to others (Vu et al. 2015).

3.1.4 Emissions Factors

The use of daily emission factors allows accurate estimation of GHG emissions from rice cultivation, but data on cultivation periods is also required. While cultivation periods can be spatially variable (Vo et al. 2020), data on lengths of cultivation periods could be obtained from interviewing farmers or from earth observation data (Torbick et al. 2017; Vo et al. 2020). Alternatively, seasonal emission factors could be used if assessing regional emissions estimates, although this would result in less refined results (Vo et al. 2020). Land use classification and GHG emissions information derived from existing literature are presented in Table 4.

Table 4. Proposed classification of key land uses within the Vietnamese Mekong Delta

Location	Land use	Conditions	GHG emissions	Reference
Red River Delta, Vietnam	Rice	General	594.0 kg CH_4-C ha-[1] yr-[1] ± 174.4 kg CH_4-C ha-[1] (SD)	Torbick et al. 2017
36 sites in Vietnam	Rice	Early season (spring season)	125-468 kg CH_4 ha^{-1} $season^{-1}$	Vo et al. 2020
		Late season (summer season)	83-1029 kg CH_4 ha^{-1} $season^{-1}$	
NW Vietnam rice cascades	Double cropped rice	Spring season, non-fertilized	97.6 kg CH_4 ha^{-1} $season^{-1}$	Oo et al. 2013
		Spring season, fertilized	48.8 kg CH_4 ha^{-1} $season^{-1}$	
		Summer season, non-fertilized	156.0 kg CH_4 ha^{-1} $season^{-1}$	
		Summer season, fertilized	95.4 kg CH_4 ha^{-1} $season^{-1}$	
Northern Vietnam	Double cropped rice with one crop maize	Spring season, mixed organic amendments	45-138 kg CH_4 ha^{-1} $season^{-1}$	Vu et al. 2015
		Summer season, mixed organic amendments	148-288 kg CH_4 ha^{-1} $season^{-1}$	
Northern Vietnam	Double cropped rice	Spring season, efficient water management, reduced residue incorporation	34.6-90.7 kg CH_4 ha^{-1} $season^{-1}$	Tariq et al. 2017
		Spring season, efficient water management, full residue incorporation	36.1-129.6 kg CH_4 ha^{-1} $season^{-1}$	
		Summer season, efficient water management, reduced residue incorporation	236.6-328.9 kg CH_4 ha^{-1} $season^{-1}$	
		Summer season, efficient water management, full residue incorporation	222.6 to 486.5 kg CH_4 ha^{-1} $season^{-1}$	
		Spring season, inefficient water management, reduced residue incorporation	98.4-123.9 kg CH_4 ha^{-1} $season^{-1}$	
		Spring season, inefficient water management, full residue incorporation	88.4-273.9 kg CH_4 ha^{-1} $season^{-1}$	
		Summer season, inefficient water management, reduced residue incorporation	148.4 to 379.5 kg CH_4 ha^{-1} $season^{-1}$	
		Summer season, inefficient water management incorporation	447.5 to 749.1 kg CH_4 ha^{-1} $season^{-1}$	
Central Vietnam lowland delta	Double cropped rice	Over both seasons	4.1 kg CH_4 ha^{-1} day^{-1}	Nauditt and Ribbe 2011

3.2 Emissions from Mangroves

3.2.1 Background

Mangrove forests are specifically found along coastal areas with tree species specially adapted to their environment with substantial woody root structures supporting the tree's growth in waterlogged soils. Mangrove systems provide multiple ecosystem services, including the dissipation of ocean wave energy (particularly valuable in minimizing the impact of extreme events such as tsunamis), the build-up of sediment over time, the absorption of pollutants, and habitat for unique fauna.

Specifically, within the VMD, the mangroves cover approximately 1,000 km^2 of area, with a linear coverage of 23%, or 3,612 km of the coastline (Global Mangrove Watch 2022). The extent of mangrove coverage within the VMD has decreased by 82 km^2 from 1996 to 2020. The following map shown in Figure 2 indicates the mangrove extent in VMD (Global Mangrove Watch 2022).

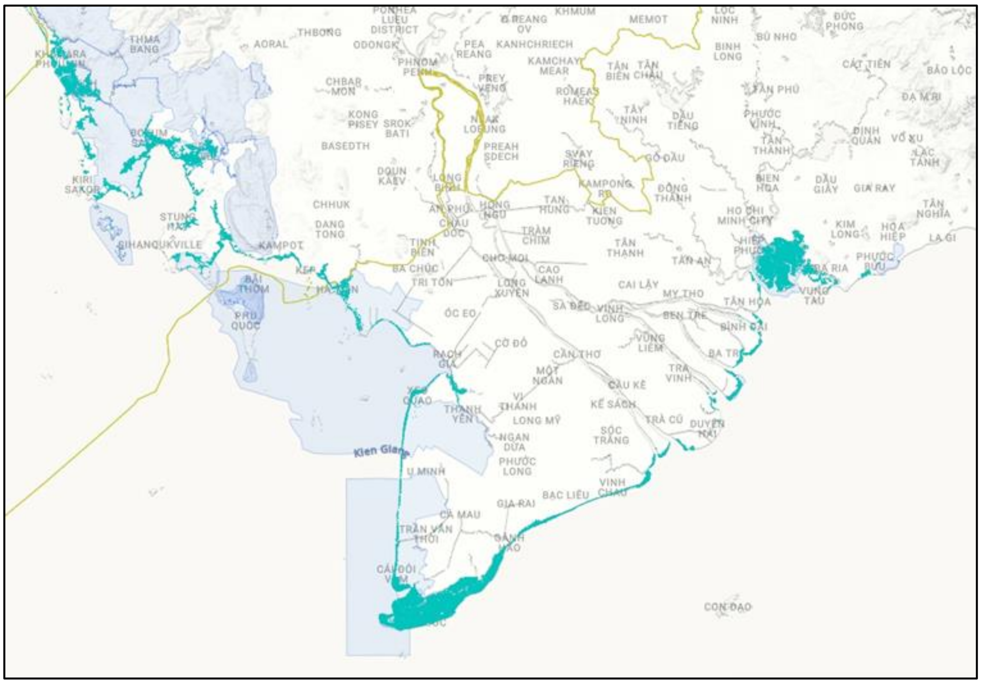

Figure 2. Map showing mangrove extent (in turquoise) in the Vietnamese Mekong Delta
Source: Global Mangrove Watch (2022)

3.2.2 Mangrove Management

Degradation of VMD's mangroves has been significant, with a 50% reduction in the mangrove forest area since the mid-1900s (Do and Thuy 2022). There are a few activities which impact on Vietnams mangroves, including:

- Extraction of wood (for timber, charcoal, or firewood)
- Seaport construction and pollution
- Shrimp Farming
- Salt Production

Seaport construction has led to the degradation of coastal ecosystems through habitat destruction and pollution of both air and water. A recent awareness of seaport impact has led to a novel approach to "green seaport development," which protects the environment while still delivering on economic development. Its development includes both the prevention of environmental damage (from pollution), as well as the conservation and enhancement of coastal ecosystems so that they can continue to provide ecosystem services and absorb and mitigate seaport impact (Tinh et al. 2022).

Vietnam is one of the world's largest producers of shrimp, with farms often found in mangrove areas. Shrimp farming has been shown to impact marine-based catch, with changes to hydrological processes, soil acidification and GHG emissions, and loss of mangrove habitat (Tue et al. 2014). Integrated, climate-resilient mangrove shrimp farming in the VMD, involving the restoration of mangroves and protection of mangrove biodiversity and ecosystems, is currently being supported via donor-based project development in partnership with the Government of Vietnam (IUCN 2016; SNV 2023). Emission estimates from shrimp farming are limited in the literature and appear to vary depending on the type of shrimp farming used. Shrimp farming methods include intensive, semi-intensive, improved extensive, mixed mangrove concurrent systems. Mixed mangrove systems have the highest CH_4 emission amount per volume of shrimp due to the land cover changes (up to 61% deforestation) for installation of this system (compared to the rice paddy system) (Järviö et al. 2022).

Human activities also cause mangrove degradation due to deforestation. Whilst mangrove wood extraction for construction and firewood is illegal in protected areas, it is a common cause of mangrove degradation by households (Vo and Kuenzer 2012).

Salt farming is concentrated in areas where seawater is most prevalent and is based on the construction of large evaporation ponds from which the salt can be collected. Installation of these ponds also results in completely removing the mangrove vegetation.

While the degradation of mangroves is an issue in the VMD, restoration in some areas has taken place, restoring both damage due to the war (Nam et al. 2016) and other developments (Tinh et al. 2020). Restoration is either done through natural regeneration or planting mixed mangrove species or monoculture. A study comparing naturally regenerated versus assisted regeneration found no significant difference in vegetative carbon stocks between the two restoration approaches (Nam et al. 2016). However, impacts of restoration on soils show a variability (potentially due to higher variability in soils) with differences reported in soil carbon stocks in the soil surface in restored mangrove forest versus intact mangrove forest (driven by lower bulk density and higher carbon content reported in the intact mangrove forest) (Tinh et al. 2020).

3.2.3 Emission Factors

There are some key influencing features to consider when exploring the GHG dynamics of the mangroves of the Vietnamese Mekong Delta.

Soil carbon

Soil carbon stocks have been shown to vary not only between locations, ranging between 186 to 1575 t C ha^{-1} in Vietnam (Järviö et al. 2022) but also at depth. A study in Northern Vietnam reported a significant variability in soil carbon at depth increments down to 100 cm (Pham et al. 2020). In addition to this, the total depth of the highly organic mangrove peat soil (and thus the potential total carbon loss from land use change) also varies from location to location.

Hydrological Influence

Hydrology is an important feature of the mangrove system-determining the rates of sedimentation, salinity intrusion and potential exposure of soils to aerobic conditions. Hydrological impact on mangroves also influences their soil condition and movement of carbon from the system, with a 50% difference reported in mangrove soils between provinces attributed to how well protected the wetlands were from waves (Tinh et al. 2020). This is also applicable to other particulates, including trace metals, whose precipitation and movement within and from mangrove systems is influenced by tidal movement (Nho et al. 2020).

3.2.4 Sequestration and emissions scenarios

A review of the current Voluntary Carbon Market Standard registries[1] returned over 50 mangrove-based carbon projects, either under development or validated by the Standards, that involve either restoration or protection of mangroves. While this demonstrates a strong potential for conservation and restoration efforts to avoid the loss of carbon volumes and sequester carbon, only limited data is available on these projects' actual carbon benefit numbers.

Carbon storage in Vietnamese mangroves has been shown to range from fringe to interior forests between 719 ± 38 into 802 ± 12 Mg C ha^{-1} [in the above and below ground biomass, woody debris and soils (Tue et al. 2014)], although this sits outside the lower bounds reported by global estimates of between 437 to 2,186 Mg C ha^{-1} and a mean value of 1043 (Nally et al. 2010). There is variability in the carbon storage in mangrove soils, depending on soil depth and organic matter contributions and the impact of coastal processes. Carbon stocks and other physiochemical properties within the soils can vary within a single mangrove system (for example, across a forest gradient), with inner forest areas exhibiting higher contributions of mangrove-contributed soil organic matter and higher carbon stocks in plant biomass than the mudflat zone (Vinh et al. 2021). Literature on above and below-ground biomass in mangroves reports considerable variability depending on location and species, while degradation of mangrove systems would result in a loss of these carbon stocks from vegetation and soils.

[1] The Verified Carbon Standard, and Plan Vivo

Mangrove restoration has the potential to sequester significant carbon, more commonly measured by the change in the woody plant material (both above and belowground biomass), as soil carbon measurements are more complex. A generalized estimate for the potential for mangrove plantations in Vietnam projected sequestration rates of up to 32 t CO_2e ha^{-1} yr^{-1} in the above and belowground biomass over a 30-year timeframe (Nally et al. 2010).

While the existing literature would not provide a better overview of the GHG emissions from mangroves, dissolved inorganic carbon from mangrove systems should also be considered when considering fluxes from these systems. Spatial variability of these has been shown (Borges and Kone 2008), particularly correlated with salinity (influenced by rainfall dilution). More rainfall and water flushing through the system may increase benthic and water column heterotrophy, as well as dissolved CO_2 originating from soil respiration, which strongly increases O_2 released to the atmosphere (Borges and Kone 2008). The impacts of climate change are therefore essential to consider on these emissions sources, particularly the increased intensity of rainfall.

Several models have been introduced based on different studies to determine the GHG emission amounts from different land uses. The applicability of these models must be identified before they are used in any study related to GHG emissions.

4. Review of models for estimating greenhouse gas emissions

4.1 Process-based Biogeochemical Models

Several process-based biogeochemical models exist that can be applied to waterlogged ecosystems (Mack et al. 2023). Some of the most suitable of these include MERES (Methane, Emissions from Rice Ecosystems), CERES (Crop Environment Resource Synthesis), MEM (Marsh Equilibrium Model), PEPRMT (Peatland Ecosystem Photosynthesis, Respiration, and Methane Transport), DNDC (Denitrification-Decomposition) and DayCent. While these models typically require site-specific data (which is beyond the focus of this review), it is important to note their potential for use (Farmer et al. 2011).

4.1.1 MERES

MERES was developed to model CH_4 emissions from rice production systems with saturated soils in tropical regions (Matthews et al. 2000). It is a process-based model of CH_4 production, CH_4 oxidation, and fluxes of CH_4 from the soil (i.e., diffusion, ebullition, plant-mediated transport) and includes the effect of O_2 concentration and alternative electron acceptors in the soil. Paddy water depth is simulated using inputs from irrigation, precipitation and outputs from drainage and evapotranspiration.

4.1.2 CERES

CERES rice simulation model is used to simulate crop growth and rhizodeposition and the influence of crop management practices (Singh et al. 1993). Overall, the model has successfully predicted seasonal patterns of CH_4 emissions associated with various rice crop management scenarios in Iran, China, the Philippines, India, Indonesia, and Thailand, which were also used for upscaling to the national level (Matthews et al. 2000; Mirakhori et al. 2017).

4.1.3 DNDC

The DNDC model is a process-based model for application at the field scale to model the decomposition and denitrification of soil under agricultural practices, delivering estimates of N_2O, CH_4 and CO_2 emissions (Li et al. 1992). Wetland-DNDC is a further development of the original DNDC model- specifically to apply on wetland sites and includes additional biogeochemical processes specifically related to soil hydrology, temperature, decomposition and CH_4 production (Zhang et al. 2002). The model requires relatively precise inputs to deliver precise outputs. Currently, the DNDC model is most applicable for forested and emergent systems, compared to the vegetation in the Mekong Delta.

4.1.4 MEM and PEPRMT

The MEM model is a mechanistic model suitable for application in salt marsh and mangrove settings and estimates aboveground biomass (AGB), belowground biomass (BGB), soil organic carbon (SOC) and surface elevation. The model was developed to describe the interactions between the physical and biological processes that govern tidal wetland response to rising sea levels and the resulting equilibrium elevation (Morris et al. 2021). The productivity outputs can then be converted to CO_2. The PEPRMT model is applicable to freshwater systems and provides estimates of CO_2 and CH_4 emissions. The two models are currently being merged, allowing a better accounting for net sequestration, but it is only applicable in non-forested emergent wetlands (both salt and fresh), and thus it is not currently applicable to a mangrove forest (Mack et al. 2023). Model application is currently limited to applications within the United States.

4.1.5 DAYCENT

DAYCENT is a biogeochemical model used to estimate ecosystem responses to changes in climate and agricultural management practices in crop, grassland, forest, and savanna ecosystems on a daily time step (Mack et al. 2023). DAYCENT uses a number of sub-models for soil water content, temperature, plant production, net primary production (NPP), nutrient cycling and gaseous emissions, including adding a component for methane emissions from saturated paddy rice soils (Cheng et al. 2013). The DAYCENT model has been applied globally and across multiple ecosystems and requires daily weather data, soil data and crop management data.

4.2 Emissions Calculators

There are several calculator-based tools available to estimate emissions from waterlogged soils, which can run on more generalized data. These include Sector, TROPP-CAT and AFOLU CARBON

4.2.1 Sector

The Sector tool[2] was developed by the International Rice Research Institute and estimates field emissions under farm management based on the IPCC methodology to estimate CH_4 emissions from rice cultivation. The tool can be applied from national to project level scenarios and allows comparison between different land management practices.

4.2.2 AFOLU Carbon Calculator

United States Agency for International Development (USAID) and Winrock International developed the AFOLU Carbon Calculator[3] and estimated the carbon impacts of project activities based on basic information shared about the project activities- location, size, and

[2] https://sector.irri.org/about
[3] https://afolucarbon.org/

management practices. The calculator uses the basic IPCC approach of combining activity data with emission factors.

4.2.3 EX-ACT

The Ex-Ante Carbon Balance Tool (EX-ACT[4]) is based on the IPCC methodology and provides estimates of agricultural interventions on GHG emissions. It covers the full spectrum of land uses under Agriculture, Forestry and Other Land Use (AFOLU) and includes inland and coastal wetlands, fisheries, and aquaculture.

4.2.4 TROPP-CAT

The Tropical Peatland Carbon Assessment Tool (TROPP-CAT) (Farmer et al. 2013) is an Excel-based tool to estimate soil carbon loss from organic peat soils. The tool is based on a function of subsidence due to drainage impact and has been applied within an Indonesian context under oil palm and Acacia plantations. A calibration would be needed for a coastal mangrove context.

[4] https://www.fao.org/in-action/epic/ex-act-tool/suite-of-tools/ex-act/en/

5. GHG emissions from potential land use change in the Mekong Delta

Considering the desk-based nature of this review, estimates of GHG emissions from the VMD under different land use classes were conducted using the tools outlined in sections 4.2.1 to 4.2.3.

5.1 Future Scenarios of Climate and Land Use Change

Climate change predictions for some parts of the Mekong Delta indicate a 1.4 °C increase by 2050 (or 0.7 °C by 2030. Additionally, there will be an increasing rainfall in rainy months up to 25% but decreasing rainfall by 30-35% by the end of the century causing the climate to be more intense in each season and resulting in an average rainfall change of 1.3% by 2030 and 2.4% by 2050 (Mackay and Russell 2011). Climatic extremes of more rainfall in the wet months and extended drier periods around the dry seasons are also predicted. Sea level is predicted to rise by 15-16 cm in 2030, to 28-32 cm by 2050 and up to 63-88 cm by 2090 (Mackay and Russell 2011).

Impacts of hydropower development along the Mekong River are anticipated to reduce sediment load in the lower stretches of the river system by 60-96%, and while the impact of a reduction in sediment load is poorly studied, it is anticipated that the long-term effects of this will influence river morphology, fish communities and access to nutrients for agriculture production, which will impact on land use and emissions from the delta (Baran et al. 2015).

5.2 Land Use Scenarios

Land use change scenarios of the delta are presented in Table 5, based on the previously identified wetland and land use types.

Table 5. Wetland land use types, anticipated change, and climate impact

Area of the delta	Wetland type	Wetland use	Potential land use change	Climate impact
Upper	Floodplain wet rice	Triple rice	Change to double rice	Increased salinity, drought
	Floodplain wet rice	Double rice		
	Floodplain wet rice	Rice and vegetable crop		
Lower	Seasonally flooded grassland	Grassland	Change to rice system	Increased salinity, inundation
	Mangrove	Aquaculture	Restoration	
	Mangrove	Degraded mangrove	Deforestation and aquaculture	
	Mangrove	Intact mangrove vegetation	Deforestation and aquaculture	

Source: Miller et al. (1999)

5.3 GHG Emissions from Rice

We used the Sector[5] model to explore management scenarios under the rice farming systems, testing differences in emissions when changing from triple to double rice crops, changes in the timing of straw incorporation, and changes to the irrigation system. Standard input variables on a per-season basis can be found in Table 6, as well as the alternative input variables used under management changes.

[5] www.sector.irri.org

Table 6. Input data use for simulations using the Sector tool.

Input	Standard scenario input variables (per season)	Management change input variables (per season)
Cultivation period (days)	105	
Harvest yield (t ha)	5	
Pre-season water regime	Two weeks non-flooded "non-flooded for less than 180 days"	
Timing of straw incorporation	Less than or equal to 30 days	More than 30 days
Amount of straw incorporated t ha	4.3 (the mean value taken from those reported in (Tariq et al. 2017)	
Other organic amendments	None	
Amount of organic amendment (t ha)	0	
In-season water regime	Irrigated/continuous flooding	Irrigated/multiple drainage periods
N fertilization rate (kg/season)	148	137 (the mean value taken from those reported in Table 1 of (Tariq et al. 2017)
Mechanized operations	Basic tillage	
Straw management	None	
Percentage of total straw used	50%	

Emissions results are found in the table below (Table 7), with the total emissions made up of both the CH_4 and CO_2 contributions. The greatest opportunity to reduce emissions was found when making changes in management practices- delayed incorporation of straw and increased drainage periods led to a more than halving of emissions in the double rice cropping system. The CH_4 emissions values presented here (Table 7) are within the range presented in the table above.

Table 7. Emissions results from simulations using the Sector tool.

Rice Cropping System	Management	Total Emissions ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	CH_4 emissions ($t\ CH_4\ ha^{-1}\ yr^{-1}$)	CO_2 emissions ($t\ CO_2\ ha^{-1}\ yr^{-1}$)
Triple rice	Standard	31.78	1.03	3.00
Double rice	Standard	21.19	0.69	2.00
Double rice	Later timing of straw incorporation	12.20	0.36	2.00
Double rice	Irrigated with multiple drainage periods	12.55	0.38	2.00
Double rice	Later timing of straw incorporation and multiple drainage periods	7.61	0.20	2.00

5.3.1 Conversion of grasslands to rice

Using the EX-ACT carbon tool (v 9.4[6]) it was estimated that over a 30-year period, the conversion of grassland to rice would give emissions of 214 t $CO_2e\ ha^{-1}$, which could be mitigated by the avoidance of this conversion. This value considers both the loss of biomass and emissions associated with the change in hydrology.

5.4 GHG Emissions from Mangroves

Global Mangrove Watch (2022) estimates the total organic carbon stored in Vietnam's wetlands at 180.13 Mt CO_2e, most of which is stored belowground in the upper 1 m of soil (158.15 Mt CO_2e). The potential for emissions mitigation within mangrove areas is estimated at just over 1,000 t $CO_2e\ ha^{-1}$ through reduced mangrove loss (assuming complete deforestation per hectare) and approximately 700 t $CO_2e\ ha^{-1}$ through mangrove restoration.

Using the AFOLU Carbon Calculator[7], 30-year simulations were run for both forest protection and forest restoration for the mangrove area of Ca Mau Province of Vietnam. Specific input values used in the model are presented in Table 8. All other input values were provided by the calculator in the geography.

[6] https://www.fao.org/in-action/epic/ex-act-tool/suite-of-tools/ex-act/en/
[7] http://afolucarbon.org/

Table 8. Input used for the AFOLU carbon calculator simulations for protection and restoration of mangroves in Ca Mau Province, Vietnam.

Input	Protection Input Variables	Restoration Input Variables
Activity type	Mangrove protection	Mangrove restoration
Forest type	Mangrove forest	Mangrove species
Avoided actions	Deforestation	NA
Effectiveness	100% effectiveness in avoiding deforestation/ illegal logging. 4 years until full effectiveness	90%

The estimated forest protection greenhouse gas benefits represent the benefits of reducing deforestation from a mangrove forest area. Forest protection activities were simulated to result in an annual average of 0.7 t CO_2e ha^{-1} (between 0.5-0.9 t CO_2e ha^{-1}) and a cumulative benefit of 21.1 t CO_2e ha^{-1} over a 30-year period for this area (Annex)- this being the gross rate of deforestation that would have been avoided. The carbon benefits can also translate into monetary benefits. Average prices for 1 t CO2e (i.e. one carbon credit) averaged USD 6.97 across the voluntary carbon market in 2023, with credits generated from land use and forestry averaging USD 11.21 in 2023 per t CO2e (Donofrio and Procton 2023).

Forest restoration activities were simulated to result in an annual average of 13.7 t CO_2e ha^{-1} (between 1.2-21.0 t CO_2e ha^{-1}) and a cumulative benefit of 412.3 t CO_2e ha^{-1} over 30 years (Annex).

6. Conclusions

There is no doubt that the VMD is a valuable natural resource for Vietnam and its people, with the considerable livelihood and ecosystem services provided by the region. Land use dynamics of the region are complex, with trade-offs and balance in decision-making needs-to consider the sustainability of use, biological value, and economic outcomes. One example of this is from the impacts noted on the use of triple-rice cropping systems, which generate additional cash in hand for farmers but, over time, require increased inputs due to loss of environmental quality of the system, and thus become more costly. Payment for ecosystem services are one option that could be developed to support transitions to sustainable land use. Design of climate mitigation activities must view the VMD through a holistic lens, with future scenarios considering the social, carbon and biodiversity benefits of activities that support sustained use and ecological functioning.

Emission scenarios vary across the delta due to differences in ecological and hydrological conditions- and this is clear from the range of rice emission values reported in Table 4. Limited data exists in the literature on the carbon storage impacts of mangrove loss or degradation and the emissions associated with conversion to aquaculture (especially shrimp farming). Further studies are required to document carbon stocks and fluxes associated with the diversity of ecosystems and land use types within the VMD- especially in the under-researched land uses such as grassland areas. A focused research strategy with coordinated research efforts can fill these gaps, and it is critical to provide decision-makers with more insight and information.

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8. Annex

Forest protection

Year	Effectiveness	Annual benefit from AD	Total benefit (t Co2e)	Cumulative benefit	Total benefit per ha	Cumulative benefit per ha
2023	100	56,290	56,290	56,290	0.5	0.5
2024	100	58,400	58,400	114,690	0.5	0.9
2025	100	60,510	60,510	175,199	0.5	1.4
2026	100	62,620	62,620	237,819	0.5	1.9
2027	100	64,730	64,730	302,549	0.5	2.4
2028	100	66,840	66,840	369,388	0.5	3.0
2029	100	68,949	68,949	438,338	0.6	3.5
2030	100	71,059	71,059	509,397	0.6	4.1
2031	100	73,169	73,169	582,566	0.6	4.7
2032	100	75,279	75,279	657,846	0.6	5.3
2033	100	77,389	77,389	735,235	0.6	5.9
2034	100	79,499	79,499	814,734	0.6	6.6
2035	100	81,609	81,609	896,343	0.7	7.2
2036	100	83,719	83,719	980,062	0.7	7.9
2037	100	85,829	85,829	1,065,891	0.7	8.6
2038	100	87,939	87,939	1,153,830	0.7	9.3
2039	100	90,049	90,049	1,243,879	0.7	10.1
2040	100	92,159	92,159	1,336,037	0.7	10.8
2041	100	94,269	94,269	1,430,306	0.8	11.6
2042	100	96,379	96,379	1,526,685	0.8	12.3
2043	100	98,489	98,489	1,625,173	0.8	13.1
2044	100	100,598	100,598	1,725,772	0.8	14.0
2045	100	102,708	102,708	1,828,480	0.8	14.8
2046	100	104,818	104,818	1,933,298	0.8	15.6
2047	100	106,928	106,928	2,040,227	0.9	16.5
2048	100	109,038	109,038	2,149,265	0.9	17.4
2049	100	111,148	111,148	2,260,413	0.9	18.3
2050	100	113,258	113,258	2,373,671	0.9	19.2
2051	100	115,368	115,368	2,489,039	0.9	20.1
2052	100	117,478	117,478	2,606,517	0.9	21.1

Forest restoration

Year	Effectiveness	Total annual benefit	Cumulative benefit	Total benefit per ha	Cumulative benefit per ha	Year
2023	90	146,316	146,316	1.2	1.2	2023
2024	90	590,770	737,086	4.8	6.0	2024
2025	90	1,075,143	1,812,229	8.7	14.7	2025
2026	90	1,513,142	3,325,371	12.2	26.9	2026
2027	90	1,877,055	5,202,426	15.2	42.1	2027
2028	90	2,160,615	7,363,040	17.5	59.5	2028
2029	90	2,367,248	9,730,289	19.1	78.7	2029
2030	90	2,504,767	12,235,056	20.3	98.9	2030
2031	90	2,582,653	14,817,709	20.9	119.8	2031
2032	90	2,610,634	17,428,343	21.1	140.9	2032
2033	90	2,597,927	20,026,270	21.0	161.9	2033
2034	90	2,552,876	22,579,146	20.6	182.6	2034
2035	90	2,482,807	25,061,952	20.1	202.7	2035
2036	90	2,394,003	27,455,955	19.4	222.0	2036
2037	90	2,291,762	29,747,718	18.5	240.5	2037
2038	90	2,180,476	31,928,193	17.6	258.2	2038
2039	90	2,063,733	33,991,926	16.7	274.9	2039
2040	90	1,944,423	35,936,350	15.7	290.6	2040
2041	90	1,824,837	37,761,186	14.8	305.3	2041
2042	90	1,706,757	39,467,943	13.8	319.1	2042
2043	90	1,591,542	41,059,485	12.9	332.0	2043
2044	90	1,480,200	42,539,685	12.0	344.0	2044
2045	90	1,373,450	43,913,135	11.1	355.1	2045
2046	90	1,271,779	45,184,915	10.3	365.4	2046
2047	90	1,175,485	46,360,400	9.5	374.9	2047
2048	90	1,084,718	47,445,118	8.8	383.6	2048
2049	90	999,509	48,444,627	8.1	391.7	2049
2050	90	919,802	49,364,429	7.4	399.2	2050
2051	90	845,471	50,209,900	6.8	406.0	2051
2052	90	776,339	50,986,239	6.3	412.3	2052

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