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Assessing the risk of utilizing tidal coastal wetlands for wastewater management

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Abstract

Coastal tidal wetlands are well recognized for the key ecosystem services they provide such as flood protection, water quality improvement, and carbon sequestration. In the southeastern United States, some communities rely on coastal wetlands for the management of secondarily treated effluents in forested and emergent wetlands. Advocates for this practice have argued that wetlands can assimilate nitrogen from wastewater, which can improve cypress-tupelo swamp productivity, and enhance marsh accretion rates to mitigate the effects of sea level rise. In contrast, evolving research on coastal wetlands and the environmental impacts of wastewater treatment pose new questions about the potential risks introduced by this practice. This review seeks to: (1) assess current research on plant productivity in fertilized coastal wetlands; (2) highlight the occurrence and fate of pharmaceuticals and personal care products (PPCPs) in municipal wastewater operations; and (3) identify knowledge gaps. Nutrient additions via wastewater augmented aboveground productivity, but decreased belowground productivity and root-to-shoot ratios. Removal efficiencies of some PPCPs by coastal wetlands have been substantial (75% - 99%), but most remain unevaluated. Furthermore, their fate and effect on local ecosystem function and biogeochemical processes remain in question. This review demonstrates that there is more research needed at both local and watershed scales to evaluate how these risk factors impact ecosystem integrity and to better understand the tradeoffs with this wastewater management practice.

Keywords

Nutrient loading; PPCPs; Antibiotics; Root-to-shoot ratio

1. Introduction

Coastal wetlands are a vital part of the natural environment and public health. They provide essential ecosystem services like water quality enhancement, carbon sequestration, and flood protection (Costanza et al., 1997; Barbier et al., 2011). In addition, coastal wetlands provide several provisioning services such as protecting endangered species, harboring commercial fisheries, and producing seafood for consumption (Craft et al., 2009). In permitted cases, these important natural resources are utilized to assimilate wastewater contaminants and to

mitigate direct discharges to nearby streams and rivers. This practice has been adopted in rural areas due to the lower costs of wastewater treatment compared to conventional means of disposal by directly discharging to streams (Breux et al., 1995; Ko et al., 2004). These systems differ from constructed wetlands and treatment wetlands which are designed, engineered, and managed for contaminant loading and treatment (Craft, 2015); they possess a wide array of ecosystem functions and wastewater additions are intended to shift these functions to favor accretion and aboveground growth (Day Jr. et al., 2004). Furthermore, the integrity of these wetlands is key to public health and ecosystem resiliency (Foley et al., 2005).

The number and characteristics of publicly owned treatment works (POTWs) that discharge wastewater into coastal wetlands for wastewater treatment is not well established. Some information is available from the National Pollutant Discharge Elimination System (NPDES) permitting database (See Supplementary Table 1; Hartwell et al., 2013; United States Environmental Protection Agency, [USEPA], 2018). These counts provide an upper limit for the number of facilities that may influence wetlands in coastal areas, but the finite number of POTWs discharging wastewater to coastal wetlands is currently unknown. In a few cases, scientists have conducted research in coastal wastewater facilities in states like Louisiana (Day et al., 2004), North Carolina (Nielsen et al., 2011; Shifflett et al., 2014; Birch et al., 2016), and South Carolina (Knight et al., 2014). These systems may be forested swamps (e.g. Birch et al., 2016; McEachran et al., 2016) or mixed marsh wetlands (e.g. Jin et al., 2010; Hu et al., 2011). Though many of these coastal wetlands are fresh water systems, many may be influenced by tidal fluctuations when established in low elevation areas and possess fluctuating salinities (Day et al., 2006; Baldwin, 2013; Stagg et al., 2017). Furthermore, continued and long-term monitoring of these systems for ecological integrity has been overlooked or has been limited to water quality impacts (Middleton and Souter, 2016). Thus, assessing the potential impacts of this practice on ecosystem resiliency requires a broad review of existing literature; and, appraisals are needed to evaluate the benefits and risks of continuing to utilize coastal wetlands for wastewater management.

The impact of wastewater loading on coastal wetlands is complex and influenced by several external forces like precipitation, local hydrology, and biogeochemical cycling. Fertilization trials have demonstrated that nutrient additions in the wetlands have improved aboveground productivity but may have mixed effects on belowground productivity (Barbier et al., 2011; Dahl, 2011; Deegan et al., 2012; Shaffer et al., 2016). Similarly, less research has evaluated the root-to-shoot ratio in these systems, and this relationship may reveal that coastal wetlands have a nutrient loading limit for maintaining ecosystem integrity (Darby and Turner, 2008a, b; Stagg et al., 2017). Thus, one objective of this review is to put the use of coastal wetlands utilized for wastewater management in the context of this evolving body of literature to assess the interaction of N-loading and biomass partitioning.

Likewise, monitoring and evaluation of pharmaceuticals and personal care products (PPCPs) in wastewater have also indicated that this practice may pose poorly understood environmental and human health risks (Brandt et al., 2015; DeVries et al., 2015; Singer et al., 2016; Christou et al., 2017). Ongoing research has demonstrated that municipal wastewater frequently contains trace concentrations of PPCPs that may pose indirect risks to

human health (Kolpin et al. 2002, 2004; Schwab et al., 2005; Lishman et al., 2006; Schriks et al., 2010; Papageorgiou et al., 2016; McEachran et al., 2017). Less attention has been dedicated to occurrence of these compounds in coastal wetlands and their food webs (Gaw et al., 2014). Furthermore, this practice has the potential to introduce endocrine disrupting compounds, persistent organic pollutants, and antibiotic-resistant bacteria in coastal wetland sediments where several food products can be derived (Cummings et al., 2010; Czekalski et al., 2012; Zhang et al., 2016). Consequently, a second objective of this research is to evaluate the state of knowledge on the occurrence and fate of PPCPs in coastal wetlands utilized for wastewater management and their potential impacts. A third and final objective of this systematic search and review (Grant and Booth, 2009; herein review) is to identify critical knowledge gaps where more research is needed to understand the tradeoffs of utilizing coastal wetlands for wastewater management.

2. The interaction of nutrient loading and biomass partitioning in coastal wetlands

The most straightforward impact of wastewater loading to coastal wetlands is the resulting change in their biomass allocation and vegetation diversity. Coastal wetlands are generally nitrogen (N) limited and nutrient additions can change the stability of the existing refractory carbon pool, the decay rate of labile organic carbon, primary production, and the allocation ratio between above- and belowground biomass production (Morris and Bradley, 1999; Hunter et al., 2009; Graham and Mendelssohn, 2010; Keim et al., 2012; Baldwin, 2013). Recent research has indicated that the impact on belowground biomass productivity can be contradictory (Deegan et al., 2012; Gregory and Mendelssohn, 2010). These efforts have demonstrated that nutrient additions can have positive, neutral, and in some cases negative implications for rhizome and root biomass. Gregory and Mendelssohn (2010) observed that nutrient loading could maintain belowground standing crop and soil matrix integrity. In contrast, Deegan et al. (2012) noted that nutrient loading decreased root and rhizome biomass, making soil matrices vulnerable to erosion by fluctuating tides. Many other authors have contributed to both sides of this debate (Valiela et al., 1976; Baldwin, 2013; Morris et al., 2013; Wigand et al., 2015). These findings expose a critical concern as rhizome and root biomass are integral to mitigating soil subsidence, decomposition, and maintaining accretion processes. As aboveground biomass and belowground biomass allocations change, the ability of the system to trap sediments and support a diversity of ecosystem services are concurrently affected (Foley et al., 2005).

2.1. Aboveground biomass productivity and wastewater loading

Research investigating aboveground plant productivity in coastal wetlands utilized for wastewater management has focused on cypress-tupelo swamps in the Mississippi Delta (Day et al., 2004; Brantley et al., 2008; Lundberg et al., 2011; Keim et al., 2012). These efforts have advocated for the use of coastal swamps and marshes for improved water quality (Brantley et al., 2008; Jin et al., 2010; Hu et al., 2011), enhanced accretion (Hesse et al., 1998; Day et al., 2004; Brantley et al., 2008), and increased growth rates for *Taxodium distichum* [bald cypress; (L.) Rich.] and *Nyssa aquatica* swamps [swamp tupelo; L.] (Day et al., 2004; Keim et al., 2012). Conclusions from this body of literature have indicated that

wastewater additions to forested swamps increased foliar turnover (Brantley et al., 2008) and improved diameter growth for trees (Day et al., 2004; Lundberg et al., 2011). Other research efforts have focused on the Carolinas (Knight et al., 2014) but are identified as managed forest plantations (Frederick et al., 1998; Ghezehei et al., 2015) and are similarly limited to aboveground growth (Shifflett et al., 2014).

Few studies have evaluated plant productivity for longer than 5 years and tend to focus on a single treatment wetland (Keim et al., 2012; Shaffer et al., 2015). Keim et al. (2012) used tree ring analysis and found that productivity decreased in a *Taxodium distichum* swamp 13 years after wastewater loading and attributed this slowed growth to increased inundation. Knight et al. (2014) monitored tree mortality and tree density ($\text{m}^2\cdot\text{ha}^{-1}$) for a Cypress-tupelo-maple swamp receiving wastewater in South Carolina, U.S. over 13 years and found that tree species less tolerant of inundation (i.e. *Acer rubrum*) became less abundant over the period of evaluation. In contrast, Hesse et al. (1998) focused on *T. distichum* swamps and found productivity increased after fertilization from wastewater loading and was sustained for 40 years of effluent discharge. These findings suggest that biological processes of coastal wetlands are sensitive to nutrient additions and hydraulic loading. However, they do not clearly demonstrate why some coastal wetlands have transitioned from forested swamp to degraded marsh while others have maintained their integrity for the duration of treatment (Shaffer et al. 2009, 2015).

Fertilization studies corroborate that adding nitrogen to these N-limited ecosystems can improve aboveground productivity, but the extent of their improved growth can be affected by salinity and tidal inundation (Davis et al., 2017; Alldred et al., 2017). These studies are abundant ($n = 99$) and show that nutrient additions improve aboveground growth across a wide spectrum of species in N-limited wetlands (Fig. 1). The most commonly studied species have been *Spartina alterniflora* ($n = 45$) followed by *Spartina patens* ($n = 9$) and *Sagittaria lancifolia* ($n = 8$). Among studies that have included fertilization, the highest productivity was found to be $1912 \text{ g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$ for *Sagittaria lancifolia* with $120 \text{ g N}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$ compared to a mean aboveground productivity of $574 \pm 539 \text{ g m}^{-2}\cdot\text{yr}^{-1}$ without fertilization (Graham and Mendelssohn, 2010, Fig. 1). The occurrence of these plant species in wetlands utilized for wastewater management is not well documented, and these rates of fertilization are typically higher than documented in other wastewater operations ($100\text{--}200 \text{ g N}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$; Day et al., 2004; Shifflett et al., 2014; USEPA, 2018).

Less abundant are evaluations of aboveground productivity for tree species in coastal wetlands. Gardner et al. (1982) measured the productivity of natural hardwood stands in the southeastern U.S. and reported productivities of $802 \pm 107 \text{ g m}^{-2}\cdot\text{yr}^{-1}$ for peat swamps, $1619 \pm 121.9 \text{ g m}^{-2}\cdot\text{yr}^{-1}$ for muck swamps, and $13,393 \pm 6774.9 \text{ g m}^{-2}\cdot\text{yr}^{-1}$ for bottomland forests. Each of these ecosystems represented unique soil moistures and organic content. Thus, productivity in these wetlands has varied based on their drainage and their soil organic content. Brantley et al. (2008) compared mean net primary productivity of a cypress-tupelo swamp receiving wastewater with similarly forested system in coastal Louisiana. Increased aboveground net primary productivity ($1202 \text{ g m}^{-2}\cdot\text{yr}^{-1}$) was observed in areas with wastewater loading relative to those without wastewater loading ($799 \text{ g m}^{-2}\cdot\text{yr}^{-1}$). These studies have indicated that increasing nutrient loading can improve aboveground growth and

thus corroborate that wastewater loading can improve aboveground primary production (Day Jr. et al., 2004; Brantley et al., 2008).

Increased aboveground productivity should be put in the context of other plant physiological and ecological processes because improving aboveground biomass does not inherently improve the resiliency of coastal wetlands. Nutrient additions have shown improved flood tolerance in wetland tree species like *Taxodium distichum* (Effler and Goyer, 2006) and counterbalanced subsidence processes for decades in the Hammond Assimilation Wetland (Day et al. 2004, 2012; Hunter et al., 2009; Morris et al., 2013). Brantley et al. (2008) documented that wastewater loading, primarily through hydraulic additions, mitigated the effects of drought and increased aboveground productivity. Many researchers have noted that these wastewater additions can be effective in buffering saltwater intrusion especially during drought years (Myers et al., 1995; McKee et al., 2004; Martin and Shaffer, 2005; Shaffer et al., 2018). In contrast to these generally positive effects, wetlands receiving wastewater for longer periods of time have demonstrated a subsequent loss of biodiversity from mortality (Conner et al., 2002; Day et al., 2004; Lundberg et al., 2011; Shaffer et al., 2015), as well as the introduction of invasive flora and fauna (Effler et al., 2006; Howard et al., 2008; Shaffer et al., 2015). Furthermore, some of these tidal freshwater wetlands convert to marsh and open water after extreme environmental stress (Shaffer et al., 2009). Thus, assessments of aboveground productivity should continue to improve our understanding of coastal wetland responses to anthropogenic nutrient additions.

Though the body of research documenting the relationship between nutrient additions and aboveground productivity generally shows that increased nitrogen loading increased productivity, more comprehensive inventories of wastewater facilities using coastal wetlands are needed to understand their efficacy as well as their relationship to coastal wetland resiliency, and whether their use for wastewater management is sustainable. Certainty around these conclusions could be improved through dedicated long-term evaluations for wastewater operations relying on natural coastal wetlands. Furthermore, such studies could improve this practice to augment ecosystem integrity and to identify concomitant events that may be influencing wetland resilience.

2.2. Belowground biomass productivity and wastewater loading

Research on the interaction of belowground productivity and nutrient additions in coastal wetlands utilized for wastewater management has not been prominent. This gap can be partially attributed to the difficulty in collecting *in situ* samples for belowground biomass, especially when considering that most facilities relying on coastal wetlands have reported the use of cypress-tupelo forested swamps. These tree root systems are dense, require excavation, and can be difficult to standardize across experiments (Vogt et al., 1998; Brunner et al., 2015). Despite the limited research in this area, further scrutiny is required to understand how plant productivity in the rhizosphere has responded to nutrient additions. High variability has been notable in fertilization studies (Fig. 1). Furthermore, the impact of nutrient additions to belowground biomass has become a contentious area of research as several authors have reported contrasting results in field scale evaluations.

Fertilization studies have primarily focused on *Spartina alterniflora* (n = 24), *Sagittaria lancifolia* (n = 13), and *Spartina patens* (n = 9; Fig. 1). These species have not been frequently documented in wastewater loading evaluations but are typical to coastal tidal wetlands and may be present after wastewater loading was initiated due to wetland degradation (Shaffer et al., 2016). These studies have documented substantial variation in the response of roots and rhizomes to nitrogen additions including decreased biomass, neutral impacts, and increased biomass. More evaluations of belowground biomass have been available for unfertilized wetlands (Fig. 1). Some of the variation in response to nutrient additions may be attributed to salt water intrusion (Stagg et al., 2017), flooding (Allen et al., 1996; Baldwin et al., 1996), or methodology (Graham and Mendelssohn, 2016). Thus, the conclusions of these research have been difficult to translate to coastal wetlands used for wastewater management. However, these studies have demonstrated uncertainty around wetland resiliency to increased nutrient additions.

Several authors have noted an interaction between nutrient loading and salinity for belowground biomass in coastal wetlands (Shaffer et al. 2009, 2016, Alldred et al., 2017; Stagg et al., 2017). In these evaluations, salts have competed with nitrogen for exchange sites and thus inhibit nutrient uptake (Seitzinger et al., 1991; Morris et al., 2013). Shaffer et al. (2009, 2016) focused on cypress-tupelo swamps and found that low salinities (<1.0 ppt) shifted forested swamps to herbaceous marshes, thus corresponding to a shift in their belowground rhizosphere composition. Despite these transitions, Shaffer et al. (2016) suggested that salinization of these wetlands did not inherently impact gross ecosystem productivity as herbaceous net primary production could compensate for the loss of tree productivity. Shaffer et al. (2009, 2016) also noted that the introduction of secondarily treated municipal wastewater could improve net primary production as well as combat salinization of these wetlands (Shaffer et al., 2009, 2016). Stagg et al. (2017) conducted a comprehensive landscape scale assessment of above- and belowground productivity across fresh (0–0.5 ppt), intermediate (0.5–5.0 ppt), brackish (5.0–12 ppt), and saline (12–20 ppt) marshes in coastal Louisiana and found that brackish marshes have both the widest range of belowground productivities and the highest productivity of all other wetland types. The most commonly studied wetland type was found to be saline wetlands with *Spartina alterniflora* (Fig. 1; Stagg et al., 2017). These studies have demonstrated that nutrient loading has an array of effects on plant productivity and coastal wetland integrity as new salinity gradients have been introduced. Simultaneously, these studies have raised the question of whether coastal wetlands will maintain their resiliency to sea level rise if their nitrogen loading is not balanced with effects of sea level rise.

Another challenge in estimating the impact of nutrient loading from wastewater to belowground biomass productivity has been the contrasting results derived from methodological approaches. Two prominent methods used to quantify belowground biomass in response to nutrient additions in coastal wetland have been the ingrowth method and the standing crop method (Graham and Mendelssohn, 2016); however, these methods have differed greatly in their conclusions (Morris et al., 2013; Graham and Mendelssohn, 2016; Stagg et al., 2017). For instance, several studies have documented that nutrient additions can improve belowground productivity as measured by living biomass (Valiela et al., 1976; Tyler et al., 2007; Hunter et al., 2009; Graham and Mendelssohn, 2016); but, these studies

frequently neglect the relative ratio of aboveground biomass to belowground biomass (Stagg et al., 2017). In contrast, other studies have demonstrated that nutrient additions reduce standing crop (living biomass + dead biomass), and thus result in reduced tensile strength in soils (Darby and Turner, 2008a, b; Ket et al., 2011; Graham and Mendelssohn, 2016). These results have led researchers to conclude that this decreased biomass response in the rhizosphere can create negative feedback mechanisms that exacerbate coastal wetland loss (McCormick et al., 2009; Deegan et al., 2012). Lastly, some studies have concluded that nutrient additions have no impact to belowground biomass (Gallagher, 1975; Buresh et al., 1980; Wigand et al., 2007; Hunter et al., 2009; Langley et al., 2009; Anisfeld and Hill, 2012). These contrasting conclusions indicate that the interaction of nutrient loading and belowground biomass in coastal wetlands is not well understood and more research is needed to understand co-factors that may be affecting productivity in the rhizosphere, and ultimately affecting ecosystem resiliency.

An emerging characteristic in considering the use of coastal wetlands for wastewater management is the effect of nutrient additions on the root-to-shoot ratio. Many authors have noted that the root-to-shoot ratio decreases as nutrient loading increases (Morris, 1982; Ågren and Ingestad, 1987; Hilbert, 1990; Ericsson, 1995; Ågren and Franklin, 2003; Darby and Turner, 2008a, b; Morris et al., 2013). This relationship has appeared to be empirically true across the reviewed efforts, but this research has been limited (Fig. 1). Morris et al. (2013) concluded that this reallocation of biomass is largely the result of establishing new equilibriums between standing root biomass and aboveground biomass. Graham and Mendelssohn (2010) reached a similar conclusion by evaluating nitrogen loading in a oligohaline marsh and additionally noted that nitrogen enrichment beyond the wetland's capacity to assimilate nutrient additions could lead to changes in ecosystem structure. Thus, a potential consequence of wastewater loading to coastal wetlands could be a slow change from a steady state plant community to a transient ecosystem that can be more readily influenced by external stressors like salt water intrusion or storm surge.

2.3. Associated risk attributed to the interaction of nutrient loading and biomass partitioning

Ecosystem productivity, biodiversity, and biogeochemical cycling in coastal wetlands are complex and intertwined. Research efforts have sought to determine if managing these critical systems with nutrient additions could augment plant productivity (both above and below-ground), increase accretion rates, mitigate sea level rise, and generally improve ecosystem resiliency. Collectively, this research has demonstrated that nutrient additions can improve above-ground productivity but has been less conclusive about belowground biomass as well as overall ecological integrity. Thus, there is some uncertainty around the impacts of utilizing natural coastal wetlands for wastewater management. Much of the research has been limited to studies conducted in Louisiana which have focused on the benefits of these nutrient additions on soil accretion processes and on improved aboveground productivity in cypress-tupelo swamps. Some wetlands used by wastewater facilities have demonstrated signs of degradation; but, without monitoring with respect to local hydrology and site conditions, it has been difficult to attribute these issues strictly to nutrient additions from wastewater. More comprehensive inventories are needed to understand how frequently states

have implemented this practice and to investigate coastal wetland integrity. Similarly, a better understanding of how nutrient additions affect belowground biomass in these critical ecosystems is needed. The relative risks of this practice may be low, based on the wastewater literature, but contextual research from fertilization trials suggests that the impacts of these nutrient additions on belowground biomass and biomass re-allocation requires further scrutiny.

3. Pharmaceuticals and personal care products in coastal wetlands

The impact of PPCPs on coastal wetlands used for wastewater management is substantially more complex and requires a holistic and interdisciplinary perspective. PPCPs have a wide range of biophysiochemical properties including, but not limited to, their molecular weight, polarities, chemical structure, degradability, and partitioning coefficients. When released into the environment, these traits result in a wide range of potential fates and the number of studies that have been able to comprehensively address these compounds are limited. Regardless, trace concentrations of PPCPs released into the environment by POTWs continue to be a concern for ecological integrity and public health.

Non-human organisms may be more sensitive than humans to trace concentrations of PPCPs in wastewater effluent and their receiving waters. For example, exposure to synthetic estrogens via municipal wastewater have been documented to collapse fish populations after repeated exposure (e.g. Kidd et al., 2007). Trace concentration of diclofenac in wastewater has also resulted in substantial declines of regional vulture populations (e.g. Oaks et al., 2004). A more recent example has included the detection of PPCPs in sediments and tissues of resident mussels near municipal wastewater outfalls (Krogh et al., 2017). Yet, few studies have documented the impact and treatment of these compounds by POTWs relying on natural coastal wetlands (Conkle et al., 2008; Conkle and White, 2012). These landscapes are important fish and wildlife habitats and have the potential to become reservoirs for human exposure to PPCPs, especially in animal species consumed by humans (i.e. shellfish).

3.1. Concentration and potential fates of PPCPs in coastal wetlands

PPCPs can have numerous fates in the environment when discharged by Publicly Owned Treatment Works (POTWs) into rivers, forest water reuse systems, and wetlands (Fig. 2). They may be transformed via microbial degradation in the rhizosphere, transferred and modified via plant uptake and phytodegradation, or they may undergo biochemical transformations due to interactions with concurring compounds and environmental conditions. These transformations can have significant impacts on chemical solubility and polarity as well as other properties that govern the environmental fate of PPCPs (Kümmerer, 2009). In addition to these transformations, PPCPs are simultaneously influenced by local hydrology where they can be diluted and transported by river flooding during high flow events or by tidal inundation due to storm surges (Fig. 2). Because many coastal wetlands provide habitat to fish and wildlife that thrive in these landscapes' substrates (Fig. 2), it is also worth considering that they may become potential reservoirs for these compounds (Gaw et al., 2014). Thus, there have been several nodes of transformation and transportation for PPCPs in natural wetlands. Because monitoring of PPCPs in natural coastal wetlands

utilized for wastewater management has been limited, most of the knowledge around the concentration and fate of these compounds comes from large POTWs and constructed wetlands used for wastewater management.

The goal of wastewater treatment is to effectively remove contaminants from their various substrates. As a result, removal efficiencies have been used as a metric of success. Conkle et al. (2008) evaluated the removal efficiencies for 15 PPCPs including cotinine, caffeine, carbamazepine, fluoxetine, atenolol, nadolol, propranolol, metoprolol, sotalol, sulfapyridine, sulfamethoxazole, acetaminophen, naproxen, ibuprofen, and gemfibrozil in a southern Louisiana wetland. By comparing inflow to wetland outflow, the assessment indicated that PPCPs were removed between 51 and >99% (Conkle et al., 2008). This research indicates that coastal wetlands may be an effective tool in reducing PPCP loading into nearby waterways. The final concentrations discharged from the wetland outfalls are within the distribution of the same compounds discharged from larger POTWs across the U.S. (see Fig. 3; Conkle et al., 2008; Hijosa-Valsero et al., 2010; Munir et al., 2011). However, the Conkle, White, & Metcalfe dataset is limited in the number of compounds evaluated and does not capture the fate of the removed compounds. Many additional PPCPs have been evaluated in POTW operations and have been found to have wide distributions in their concentration (Fig. 3). Furthermore, the dataset represented a low sample number for each compound ($n = 3$) and did not address seasonality of PPCPs in wastewater influents and removal efficiencies. These data gaps are essential to understanding PPCPs in wetlands and their potential impacts to ecosystem resiliency. More direct assessments of these facilities are needed to understand the associated risk of this practice.

Evaluations outside of coastal wetlands have demonstrated that PPCP concentrations from POTWs can vary by season, region, and similarly partition between water matrices, sediments, and biota (Kümmerer, 2009; Hijosa-Valsero et al., 2010; Hughes et al., 2013; Gaw et al., 2014; Chen et al., 2016; McEachran et al., 2017). These evaluations have provided valuable insights to what PPCP have been detected, the potential physiochemical conditions that have influenced PPCP, and have demonstrated the many challenges in understanding how these compounds persist and transform in the environment. Gaw et al. (2014) reviewed published literature on the PPCPs in marine environments and concluded that the most common compounds reported were antibiotics and their metabolites (41 different compounds); though, non-steroidal anti-inflammatory drugs (NSAIDs) like ibuprofen also tend to be commonly detected (Hughes et al., 2013; Chen et al., 2016). Hughes et al. (2013) conducted a critical review on freshwater systems and noted that PPCPs may have ecotoxicological impacts on organisms due to chronic, long-term exposure; but, there are substantial experimental design challenges in capturing those impacts, as they can influence anything from reproduction and gestation to organ function (Hughes et al., 2013). Chen et al. (2016) evaluated constructed wetlands for PPCP removal efficiency and identified that some compounds (ibuprofen and paracetamol) occurred at higher concentrations during colder months, likely due to differences in human consumption. In contrast, antibacterial agents (i.e. triclosan) occurred at higher concentrations during warmer months. Thus, evaluating the impact of PPCPs in coastal wetlands requires a broad scope to understand not only what removal efficiencies exist but also to determine the fate of the compound and how that fate may change through time.

Endocrine disrupting compounds (EDCs) are a suite of PPCPs that have recently come under greater scrutiny as they have been shown to produce adverse developmental, reproductive, neurological, and immune effects in both humans and wildlife. Vymazal and Kröpfelová (2008) summarized that many EDCs classes exist (phthalates, pesticides, polychlorinated biphenyls, dioxins, polycyclic hydrocarbons, alkyl phenols, bisphenols, and steroid estrogens), and they occur frequently in most wastewater streams (municipal, agricultural, and industrial). Research efforts have demonstrated that these compounds affect the reproductive processes of freshwater and marine organisms; they can also lead to feminization of male fish (Purdom et al., 1994; Rodgers-Gray et al., 2000; Tan et al., 2007; Vymazal, 2009; Tolussi et al., 2018). Assessments for these compounds in coastal wetlands used for wastewater management have been largely overlooked and this oversight is particularly concerning for POTWs that operate in and around areas where fish may be harvested for human consumption. Constructed wetlands have demonstrated these compounds can be removed with high efficiency (>65%–99%; Vymazal, 2009); thus, coastal wetlands may also be effective in removing EDCs. However, these conclusions do not reflect how these compounds affect local fauna and cannot provide insight about the use of coastal wetlands for wastewater treatment increases risk for human exposure through recreational use of those waters or through consuming harvested food products.

More research on the occurrence and fate of PPCPs is critical to understanding the potential risks of using coastal wetlands for wastewater management. The growing body of literature has shown that these compounds can have multiple fates and can be influenced by local and regional factors as well as climates. Wetlands may be effective at removing these compounds, but comprehensive and repeated measures are needed to better understand whether use of coastal wetlands for wastewater management is useful for removing PPCPs. Similarly, the tradeoffs of this practice have not been well characterized and more efforts are needed to understand how discharging wastewater to natural coastal wetlands may create new pathways for human exposure to these contaminants.

3.2. Antibiotics and their impacts on ecosystem function and public health

Antibiotics and antibiotic-resistant genes are a developing area of concern for PPCP in the natural environment. Antibiotics are present in municipal wastewater due to their incomplete metabolism in the body as well as direct disposal of unused or expired medications (Cummings et al., 2010; Rizzo et al., 2013; Brandt et al., 2015; DeVries et al., 2015; Grenni et al., 2018). The environmental impact of these medicines is has not been well characterized, but many scientists have documented that their presence has exacerbated the risk for proliferating antibiotic-resistant genes and bacteria (Cummings et al., 2010; Rizzo et al., 2013; Brandt et al., 2015; Singer et al., 2016), created new risks to human health (Rizzo et al., 2013; Pepper et al., 2018), and impacted biogeochemical cycling (Conkle and White, 2012; DeVries et al., 2015; Yin et al., 2017). Similarly, studies have documented that the presence of antibiotics alters the biodiversity of microbiomes critical to ecosystem function (Cummings et al., 2010; Brandt et al., 2015; Grenni et al., 2018). Research on antibiotics, antibiotic-resistant bacteria, or microbial diversity in coastal wetlands used for wastewater management is limited (Conkle et al., 2008; Conkle and White, 2012). This section reviewed

the occurrence of these micropollutants in wastewater effluents, as well as the potential environmental impacts relevant to coastal wetlands.

Conkle et al. (2008) evaluated two sulfonamides (Sulfapyridine and Sulfamethoxazole) in a natural wetland in coastal Louisiana and found substantial concentrations of the latter ($>4000 \text{ ng L}^{-1}$) in POTW inflow. Wastewater residence time in the wetland was estimated as 27 days, and the treatment system reduced sulfamethoxazole concentrations by 91.3% (350 ng L^{-1}). Other common antibiotics in wastewater inflows include ofloxacin, trimethoprim, and sulfamethazine (Fig. 2; Kostich et al., 2014). Conkle and White (2012) later screened the effects of antibiotics (ciprofloxacin, sulfamethoxazole, and tetracycline) on microbial respiration in wetland soils and found significant impacts at environmentally relevant concentrations ($<1000 \text{ ng L}^{-1}$ – $50,000 \text{ ng L}^{-1}$). At these low concentrations, methane (CH_4) and carbon dioxide (CO_2) production increased while at higher concentrations microbial respiration decreased (Conkle and White, 2012). This result indicated that trace concentrations of antibiotics like sulfamethoxazole could increase greenhouse gas emissions from these coastal wetlands, but more research is needed to monitor and evaluate soil respiration from these landscapes.

Research outside of natural wetlands has similarly demonstrated that sulfonamides and other antibiotics can have negative impacts on biogeochemical cycling of carbon and nitrogen in soils (Hou et al., 2015; DeVries et al., 2015; Yin et al., 2017; Grenni et al., 2018). DeVries et al. (2015) reported on low concentration antibiotic effects on nitrate (NO_3) and nitrous oxide (N_2O) flux from sandy and sandy loam soils in incubation chambers. NO_3 removal was significantly increased after $\sim 30 \text{ h}$ exposure to 1 ng L^{-1} sulfamethoxazole, denitrification increased by 40% after 24 h exposure to the same antibiotic, and N_2O emissions significantly increased ($p = 0.0067$) after 96 h exposure to 10 ng kg^{-1} of Narasin. In contrast, Hou et al. (2015) found a decrease in denitrification rates by inhibiting denitrifying bacteria with increasing concentrations of sulfamethazine, but similarly found an increase in N_2O emissions at 5000 ng L^{-1} . Grenni et al. (2018) completed a literature review and found a range of effects of antibiotics on the nitrogen cycle (see Table 1). The impacts of these results have been concerning as coastal wetlands utilized for wastewater management receive wastewater inputs daily ranging from 0.6 to 8.0 million gallons per day.

A direct public health concern from antibiotics in natural wetlands is the impact and horizontal transfer of antibiotic-resistant genes in bacteria and wetland microbes (Murray et al., 1984; Cummings et al., 2010; Munir et al., 2011; Rizzo et al., 2013; Brandt et al., 2015; Hou et al., 2015; Singer et al., 2016; Christou et al., 2017; Grenni et al., 2018; Pepper et al., 2018). Grenni et al. (2018) reviewed the impacts of antibiotics on ecotoxicological endpoints and classified them as both direct (microbial community structure change, microbial biomass, pollution-induced community stress) and indirect (modification of bacteria ecology, resistance development, and pharmaceutical biodegradation). Brandt et al. (2015) reviewed the impact of antibiotics to ecosystem services and similarly noted the need for assessing endpoints where environmental subcomponents (i.e. microbe biomes, worms, mollusks) may be impacted. Cummings et al. (2010) evaluated plasmid-mediated transfer of antibiotic-resistant genes and concluded that wetlands receiving wastewater have the potential to be reservoirs of antibiotic genes. These findings raise concerns about food

products sourced from coastal wetlands and surrounding areas that may have increased exposure to antibiotics. That risk may be amplified when wastewater effluents are treated with chlorine (Murray et al., 1984; Munir et al., 2011; Singer et al., 2016).

Collectively, these studies and reviews suggest that antibiotic loading to natural wetlands via wastewater discharges has various potential impacts to wetland ecology and public health. More robust research is needed to understand how these natural resources are being impacted by wastewater loading. Furthermore, more field evaluations are needed to assess if these problems can be detected at the watershed scale.

3.3. Associated risk attributed to PPCPs

The effect of PPCP loadings to ecosystem integrity of coastal wetlands utilized for wastewater management has been unclear, especially when considering the multiple other stressors concurrently affecting ecosystem function (i.e. extreme weather, sea level rise, invasive species). However, several areas of concern have been identified. Antibiotics at trace concentrations may be increasing greenhouse gas emissions from these landscapes and inhibiting the denitrification process. They may also create new reservoirs for antibiotic resistance. Research efforts have sought to evaluate if these occurrences are harmful to human health, but equal attention should be given to ecosystem integrity as these wetlands can be essential ecosystem service providers. In all cases, there has been little comprehensive data for conclusive evaluation, and more assessments are needed to understand PPCP occurrence, fate, and impact to natural wetlands. Furthermore, these evaluations need continuous datasets rather than grab sampling to understand how PPCPs discharges change through time. The relative risks of discharging wastewater to coastal wetlands may be low, but contextual research from other POTW operations suggests that these practices warrant further consideration of possible risks such as loss of belowground biomass and PPCP impacts.

4. Conclusions

This risk assessment sought to evaluate potential impacts from discharging municipal wastewater into natural coastal wetlands. Nutrient additions and their impact on biomass allocation were considered as a potential issue for these wetlands, and some fertilization trials have demonstrated problematic shifts in overall biomass allocation. PPCPs were also a concern for these systems as their release has unknown fates and could result in potential pathways for human exposure. Overall, more research is needed to understand the ecology of these systems and how discharging wastewater into natural wetlands affects ecosystems services and ultimately human health.

Wastewater loading to wetlands can provide nutrient amendments and augment aboveground growth and provide solids for accretion. However, these additions may simultaneously lead to an imbalance between aboveground and belowground biomass as the root-to-shoot ratio decreases. Substantial research has been conducted on aboveground growth in these critical ecosystems and suggests that the practice is effective in improving plant growth. However, these practices have other consequences. Much less evidence is available for how the practice affects belowground biomass, but fertilization studies suggest nutrient additions

may exceed plant capacity for uptake and lead to unbalanced root-to-shoot ratios. Similarly, increasing nutrient loads may shift these ecosystems from forested to herbaceous marshes and create ideal environments for invasive flora and fauna. Evaluations are needed in these landscapes to determine if fertilization studies are indicative models for coastal wetlands receiving wastewater.

PPCP in coastal wetlands used for wastewater management has been a concern and their impact remains unclear. More evaluations considering the multiple fates of PPCPs are needed in coastal wetlands utilized for wastewater management. Trace concentration of antibiotics have been documented to occur in POTWs and some research has shown these compounds to impact biogeochemical cycling of nitrogen and carbon. These substances have not been comprehensively evaluated in coastal wetlands used for wastewater management, but their occurrence and proximity to significant fish and wildlife habitat warrants investigations to understand the associated risk.

A major conclusion of this appraisal is that better evaluations are needed for POTWs discharging to wetlands for wastewater management, and should consider multiple stressors as they are at the frontier of salt water intrusion, extreme weather, and invasive species. It is important as innovative methods develop and scientific knowledge expands that these practices are re-evaluated to ensure their intended purpose is being met and protected to maintain their integrity and ensure that both wildlife and human health is protected. Given the types of risk affiliated with the practice, more scrutiny should be applied when determining if these POTWs are sustainable and whether these wetlands have the capacity to endure the stress of wastewater loading while mitigating other external stressors.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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Fig. 1.

Scatterplots for mean aboveground productivity, mean belowground productivity, and estimated root-to-shoot ratio versus nitrogen loading for wetland plant species inventoried by Morgan (1961); Cahoon (1975); Kirby and Gosselink (1976); Valiela et al., (1976); White et al., (1978); Linthurst and Reimold (1978); Chalmers (1979); Gallagher et al., (1980); Shew et al., (1981); Birch and Cooley (1982); Roman and Daiber (1984); Hardisky et al., (1984); Gordon et al., (1985); Schubauer and Hopkinson (1984); Dame and Kenny (1986); Giroux and Bedard (1988); White and Simmons (1988); Kaswadji et al., (1990); Pezeshki and DeLaune (1991); Benito and Onaindia (1991); da Cunha Lana et al., (1991); Callaway and Josselyn (1992); Daoust and Childers (1998); Ibañez et al., (1999); Scarton et al., (2002); Brantley et al., (2008); Darby and Turner (2008a, b); Graham and Mendelssohn (2010); Stagg and Mendelssohn (2010); Ket et al. (2011); Baldwin (2013); Graham and Mendelssohn (2014); Graham and Mendelssohn (2016); Davis et al. (2017); Stagg et al., (2017). Grey areas represent nitrogen loading rates typical to wastewater (WW) facilities utilizing coastal wetlands. Tabular data provided in Supplementary Material Table 2.

Fig. 2. Conceptual framework for the life cycle of pharmaceuticals and personal care products (PPCPs) from wastewater treatment to their potential fate in natural coastal wetlands. Solid lines are direct relationships and dashed lines are indirect relationship.

Fig. 3.

Box and whisker plots showing the distribution of PPCPs from 50 POTW facilities from around the US adapted from Kostich et al., (2014) and mean concentrations PPCPs (n = 7) detected in wetlands utilized for wastewater (WW) management adapted from Conkle et al., (2008).

Table 1.

Impact of various antibiotics on the nitrogen cycle. Adapted from Grenni et al. (2018).

Process	Effect	Class of Antibiotics	Conc	Exposure (days)	Reference
Respiration	Decrease	Quinolones and Fluoroquinolones (Ciproflaxin)	0.5–2 mg mL ⁻¹	4	Heuer et al. (2008)
	Inhibition (75%)	Aminoglycosides (Streptomycin)	400 mg L ⁻¹	0.08–0.17	Weber et al. (2014)
Nitrification	Decrease	Sulfonamide (Sulfadiazine)	10–100 mg kg ⁻¹	32	Tomlinson et al. (1966)
	Decrease	Sulfonamide (Sulfadimethoxine)	50–200 mg kg ⁻¹	50	Kotzerke et al. (2008)
	Inhibition (≈25%)	Sulfonamide (Sulfadiazine)	100 mg kg ⁻¹	32	Toth et al. (2011)
	Inhibition (50%)	Tetracycline (Oxytetracycline)	12.5–75 mg L ⁻¹	7	Klaver and Mathews (1994)
	Inhibition	Tetracyclines (Chlortetracycline)	50–200 µg kg ⁻¹	50	Ahmad et al. (2014)
	Inhibition	Tetracycline (Chortetracycline)	1 mg L ⁻¹	5	Underwood et al. (2011)
Denitrification	Decrease (47%)	Sulfonamides (Sulfamethoxazole)	1 µg L ⁻¹	19	Xu et al. (2016)
	Decrease	Sulfonamides (Sulfamethazine)	100 ng L ⁻¹	2.08	Hou et al. (2015)
	Decrease	Glycopeptides (Vancomycin)	1 mg L ⁻¹	24	Li et al. (2011)
	Decrease (10%)	Tetracycline (Oxytetracycline)	5.5–7.35 mg kg ⁻¹	7	Kumar et al. (2005)