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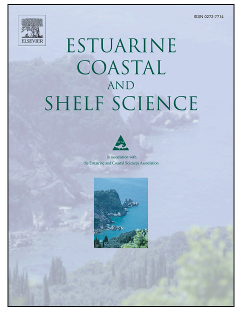

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Evaluating trade-offs of a large, infrequent sediment diversion for restoration of a forested wetland in the Mississippi delta

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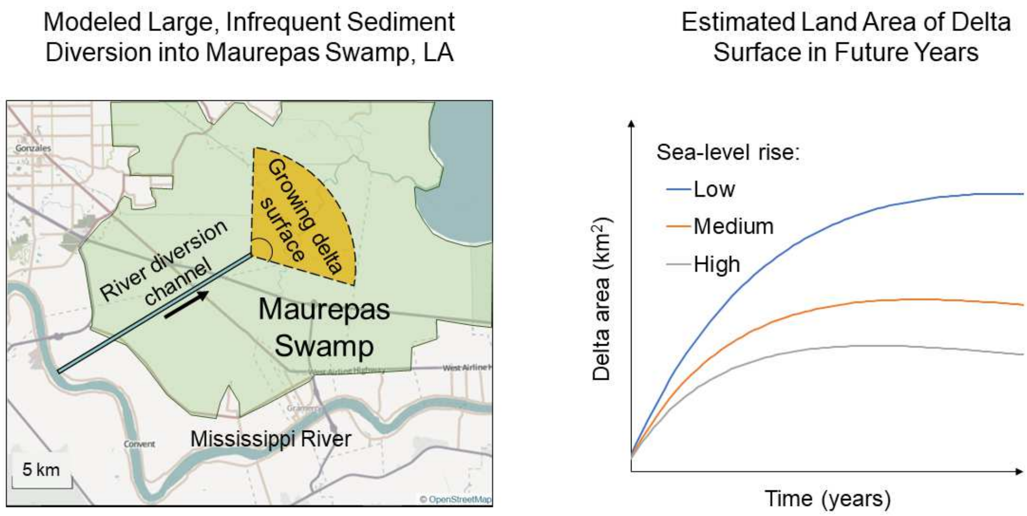
Graphical Abstract:



Abstract:

Flood control levees cut off the supply of sediment to Mississippi delta coastal wetlands, and contribute to putting much of the delta on a trajectory for continued submergence in the 21st century. River sediment diversions have been proposed as a method to provide a sustainable supply of sediment to the delta, but the frequency and magnitude of these diversions needs further assessment. Previous studies suggested operating river sediment diversions based on the size and frequency of natural crevasse events, which were large (>5000 m^3/s) and infrequent (active < once a year) in the last naturally active delta. This study builds on these previous works by quantitatively assessing tradeoffs for a large, infrequent diversion into the forested wetlands of the Maurepas swamp. Land building was estimated for several diversion sizes and years inactive using a delta progradation model. A benefit-cost analysis (BCA) combined model land building results with an ecosystem service valuation and estimated costs. Results demonstrated that land building is proportional to diversion size and inversely proportional to years inactive. Because benefits were assumed to scale linearly with land gain, and costs increase with diversion size, there are disadvantages to operating large diversions less often, compared to smaller diversions more often for the immediate project area. Literature suggests that infrequent operation would provide additional gains (through increased benefits and reduced ecosystem service costs) to the broader Lake Maurepas-Pontchartrain-Borgne ecosystem. Future research should incorporate these additional effects into this type of BCA, to see if this changes the outcome for large, infrequent diversions.

Highlights:

- Land building is proportional to sediment diversion size

- Land building is inversely proportional to sediment diversion time inactive
- Based on benefits of land gain only, large infrequent diversions are disadvantaged
- A wider set of ecosystem services would benefit BCA for large infrequent diversions
- Future work should look at BCA implications of basin-wide benefits

Keywords: wetland, coastal restoration, climate change, cost-benefit analysis, Louisiana, Mississippi delta

1 Introduction

During the 20th century, Louisiana lost about 25%, or 4800 km^2, of coastal wetlands, due mainly to the effects of human activities (Couvillion et al., 2011). One of the major causes is leveeing of the Mississippi River (MR) and its distributaries, which has isolated deltaic wetlands from the MR, preventing overbank flooding and crevasse formation (Day et al., 2000, 2007, 2016a). Engineered sediment diversions, which divert sediment and nutrient laden freshwater from the MR to adjacent wetlands, have been identified as a critical tool in restoring the Mississippi river delta plain (MRDP) (Day et al., 2007, 2016a; Kim et al., 2009; Allison and Meselhe, 2010; Paola et al., 2011; CPRA, 2012, 2017; Dean et al., 2014; Wang et al., 2014). Three operational river diversions were constructed for the purpose of restoration: the Caernarvon and Davis Pond diversions (99 and 302 m^3/s, respectively) control salinity intrusion, and the West Bay diversion (566 m^3/s) is designed to divert sediment to create and nourish wetlands near the mouth of the river. The Bonnet Carré spillway (operated at 3000-9000 m^3/s, several weeks to two months at a time every five to seven years on average), although intended for flood control rather than restoration, has led to a highly sustainable forested wetland adjacent to Lake Pontchartrain (Day et al., 2012).

If the MRDP's historic functioning is used as a blueprint for restoration, much bolder action is required (Condrey et al., 2014; Day et al., 2016a). Saucier (1963) and Davis (1993, 2000) documented numerous crevasses along the lower MR prior to major anthropogenic alteration. For example, the Bonnet Carré crevasse functioned intermittently in the second half of the 19th century with discharge ranging from 2000 to 6500 m^3/s and built a crevasse splay of about 70 km^2 as well as filling in parts of western Lake Pontchartrain with up to 2 m of sediment (Saucier, 1963; Davis, 1993) (a "crevasse splay" is defined as a fan-shaped deposit of sediment formed when a river spills water and sediment over or through a break in the river levee). Also, the 1927 artificial crevasse at Caernarvon resulted in a crevasse splay of about 130 km^2 with sediment deposition as high as 40 cm in only three months (Day et al., 2016b). Day et al. (2016a) presented the concept of large (>5000 m^3/s) and infrequent (active < once a year) diversions that would replicate the size and frequency of historic river crevasses. They hypothesized that, compared to an annually operated diversion, an infrequently operated diversion would still provide ample sediment for land building but with substantially lower impacts on water levels, salinity, nutrient load, and fisheries – controversial effects that have impeded implementation of diversions (Caffey and Schexnayder, 2002; Day et al., 2016a).

Maintaining land in the MRDP's lower reaches is becoming increasingly difficult, due to both a reduced sediment load in the MR (from dams and land use change in the upper basin), and accelerating eustatic sea-level rise (SLR) (Pfeffer et al., 2008; Blum and Roberts, 2009; Horowitz, 2010; Meade and Moody, 2010; Parris et al., 2012; Giosan et al., 2014). Also, given that subsidence generally decreases moving from the delta terminus upriver (Zou et al., 2016), land building should be more sustainable in the upper, more inland reaches of the delta. One potential location for a sediment diversion in the MRDP upper reaches is the Maurepas swamp, a 57,000 ha baldcypress-water tupelo (*Taxodium distichum - Nyssa aquatica*) forested wetland system located in the western Lake Pontchartrain Basin between Baton Rouge and New Orleans, Louisiana. The swamp is currently on a trajectory towards open water and the causes are numerous but well known; the dominant issue is that sediment and freshwater inputs from the MR that nourished the wetlands during seasonal flooding events in the past are now prevented by flood control levees (Shaffer et al., 2003, 2009, 2016; Keddy et al., 2007; Day et al., 2012).

Modelling of sediment diversions is important to understand performance and trade-offs of different operation approaches. The simplest models predict land gain based on mass balance and a uniform geometry (e.g. Parker et al., 1998; Dean et al., 2012, 2014), whereas more complex models simulate the physics of fluid flow and sediment transport based on basin hydrology and bathymetry (e.g. Edmonds and Singlerland, 2007). Predicting future scenarios with such modelling tools is useful in combination with benefit-cost analysis (BCA), where the economic benefits of different project options are compared to the economic costs, traditionally in monetary terms. For example, Kenney et al. (2013) combined a land building model with a cost model to assess trade-offs of cost, land building, and water usage for portfolios of sediment diversion projects. They used a physical unit (area of land built) to express benefits, where in other studies benefits are sometimes expressed in ecological terms, such as habitat suitability indices (Bartoldus, 1999; CPRA, 2012, 2017). However, for policymakers and politicians, who are used to making decisions with dollars, such biophysical units are less intuitive.

The "ecosystem services" framework has recently gained traction as a means for communicating the benefits of natural systems. Especially in the management of coastal systems, which provide a rich array of benefits under increasing strain from human development (Turner and Schaafsma, 2015). Ecosystem service valuation (ESV) offers a means to capture, in monetary terms, these benefits. ESV is especially useful in BCA, where benefits and costs can be expressed with a common unit, but there exist methodological challenges which make its application difficult. In the 2012 Comprehensive Master Plan (CMP), for example, the Coastal Protection and Restoration Authority (CPRA) avoided representation of ecosystem services in monetary terms, stating that "we did not include this economic aspect of ecosystem services in the master plan analysis [because] [m]odels to analyze this aspect were not readily available, and we did not have time to develop them ourselves” (the same approach is taken in the 2017 CMP). Recent examples of combined modelling and ESV exercises applied to ecosystem restoration exist in the MRDP and Florida Everglades (Mather Economics, 2010; Caffey et al., 2014; REC & EE, 2016).

This study explored a large, infrequent sediment diversion, of the sort described by Day et al. (2016a), into the Maurepas swamp (unless otherwise stated, by “diversion” we mean “sediment diversion”, a diversion intended to build land, versus a “freshwater diversion” which is intended to control salinity). Day et al. (2016a) suggested that large diversions operated infrequently are advantageous to small diversions operated continuously, but lack a quantitative assessment of the drawbacks of infrequent operation. In particular, what are the drawbacks of “curtailing” sediment delivery for one year or more compared to continuous operation? This paper addressed this question. By parameterizing a delta progradation model for the Maurepas swamp we were able to estimate land building for a number of diversion sizes, operation strategies (years inactive between operations), and SLR scenarios. First, we analyzed the relationship between years inactive, size, and land building in general, and assessed the potential to sustain land building. Second, we used the land building estimates to further assess large, infrequent diversions by performing a BCA, where ESV is applied to capture, in monetary terms, the benefits provided by the Maurepas swamp restoration. Based on the applied model, our estimates of ecosystem service benefits were limited to those provided by subaerial land (above the water surface). In addition to land building, diversions (both sediment and freshwater) also have far-reaching impacts on habitat and water quality. Though not quantified in this study, linkages between diversions and these secondary effects are discussed.

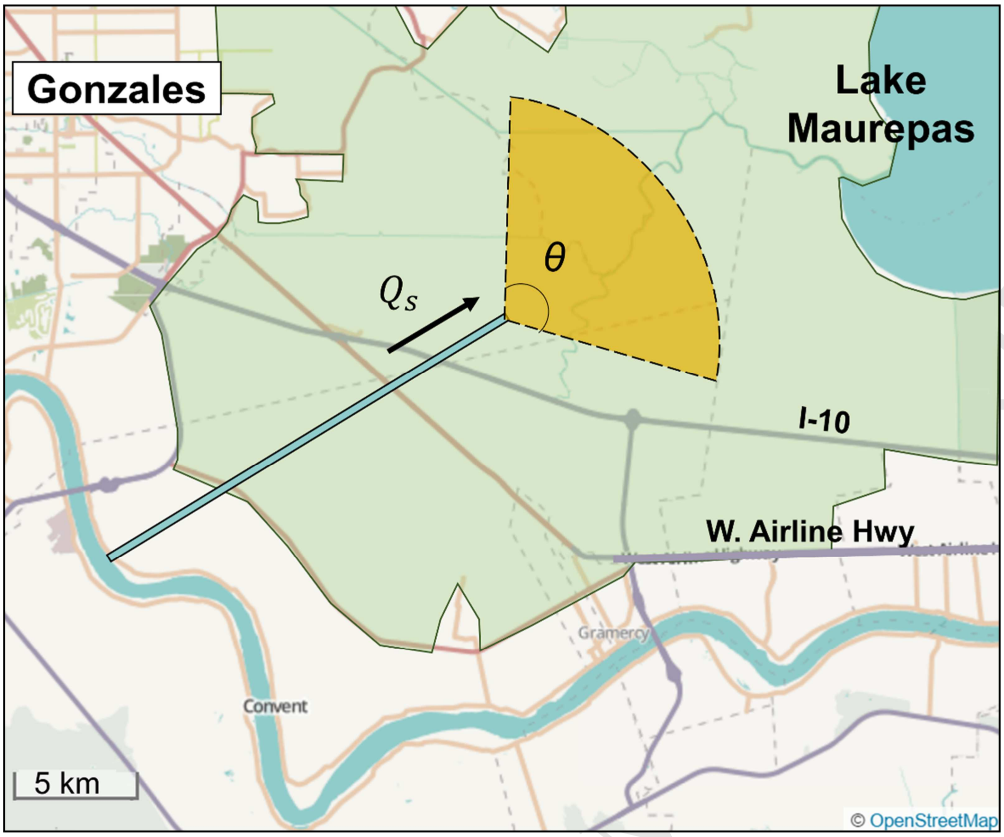


Figure 1: Conceptual diagram of a sediment diversion into the Maurepas swamp (top view) with sand discharge, Q_s, and fan spreading angle, θ. Light green areas are wetland, and light grey areas are developed.

2 Methods

2.1 River diversion modelling

To model land gain from the diversion, we used a delta progradation model (henceforth, referred to as the DPM) originally developed by Parker et al. (1998). The DPM is spatially averaged and describes the transport and deposition of sand according to the Engelund-Hansen formulation (Engelund and Hansen, 1972). The DPM has been used by Kim et al. (2009) to successfully match the growth trajectory of the Wax Lake Delta. Here, we describe the basic parameterization of the model for the Maurepas swamp in terms of the sediment input and basin characteristics (for a technical description of the DPM see Parker 2016).

2.1.1 Sediment input parameterization

The sediment input to the DPM is defined by the fractions of sand and mud available in the river, the timing and duration of the diversion pulse, and losses in the engineered guide channel. Sediment availability in the river is a function of river discharge, described here with a 12-year average hydrograph (October 2004 – September 2016, see SI-A.1.2), according to separate sand and mud (larger and smaller than 62.5 μm, respectively) rating curves. A fraction of this water and sediment is periodically captured by the diversion according to its size (discharge capacity) and frequency of operation. To be clear, although the stated purpose of this paper was to compare infrequently operated diversions against continuously operated diversions, the difference between the two strategies is not definite. All diversions must be shut off (or curtailed) at certain times during the year due to the sensitivity of vegetation and wetland species to temperature and salinity changes, and also river management issues (such as ensuring a sufficient river stage for navigation). Therefore, we defined diversion operation frequency according to two variables: X, the fraction of time the diversion is active during an operation year, and P, the number of years of inactivity in between operations. This study defined an infrequently operated diversion as one with P>0, and a continuously operated diversion as one with P=0. All tests in this study were assigned X=0.25, according to a best-practice operation strategy recommended by Peyronnin et al. (2017) (see SI-A.1.2).

An engineered guide channel is required to capture water and sediment from the MR and direct it towards the diversion outfall location. Although a deeper diversion channel would require larger abutments and a more expensive gated control structure, suspended sediment is most available at greater depths and decreases substantially with distance from the river bottom (Nittrouer et al., 2011). We assumed that at all discharge levels, the depth of the diversion channel is 24 m. At this channel depth, based on the depth of the river and sediment profile, ~30% of the sand fraction is available to be diverted (see SI-A.1.2). Once sediment is captured by the channel, it also must be flushed through the channel to the swamp. For this study, we did not model sediment transport within the guide channel, therefore, any gains or losses in transport efficiency with changing cross-sectional area were not accounted for in this model.

2.1.2 Basin parameters

Basin parameters for the Maurepas swamp were determined mostly using information available on the Coastwide Reference Monitoring System (CRMS, 2017) (basin slope, water depth, shallow soil compaction), and, if no data were available, appropriate analogs at different locations were used (mud retention) (see SI-A.1.4 for assumptions, summarized in Table A2). Three scenarios for relative SLR, defined as the sum of local subsidence and eustatic (global) SLR, were developed to cover the range of possible trajectories. Scenarios for eustatic SLR used in this study were developed to be consistent with environmental scenarios in the 2017 CMP (Meselhe et al., 2017). Correcting for the 8 cm of SLR that has occurred in the Gulf of Mexico between 1992 and 2016 (NOAA STAR, 2017), the low, medium and high scenarios correspond to 0.92, 1.42, and 1.90 m SLR from 2016 to 2100 (see SI-A.1.3 for mathematical derivation). Low, medium, and high estimates for subsidence are 1.5, 6, and 10.5 mm/year, respectively based on estimates obtained through a range of methodologies (see SI-A.1.3, Penland and Ramsey, 1990; Meckel et al., 2006; Jankowski et al., 2017; Nienhuis et al., 2017).

Along with the averaged basin parameters, sediment transport was modeled using the Engelund-Hansen formulation. We used the Engelund-Hansen calibration parameters derived by Kim et al. (2009), who calibrated the DPM's sediment transport formulation to land building in the Wax Lake Delta.

2.2 Estimating habitat change

Land area values alone are not sufficient for combining the DPM with ESV. First, we needed to estimate habitat change with and without restoration. This was required because ecosystem service values may differ between different habitat types and also because we were interested in the net benefit of restoration, or the improvement in habitat quality beyond that in a future without action (FWOA). We represented the Maurepas swamp with three habitat types: swamp (forested wetland), marsh (herbaceous emergent wetland), and open water.

FWOA simulations, which estimate the average proportion of habitat types in the Maurepas swamp, were based on projections from the 2017 CMP data viewer (CPRA, 2017b), and modified slightly according to expert opinion (projections were amended such that marsh disappears sooner than in the CMP projections - SI-B) (Figure 2). The 2017 CMP data viewer projects habitat change using the same eustatic SLR scenarios used for the DPM. To estimate habitat change with a river diversion, we assume subaerial land built in the DPM created new swamp habitat. Although there would be a time lag between the deposition of sediment and recruitment of cypress and tupelo trees, the assumption is still reasonable given that the balance between sediment accretion and relative SLR is the dominant forcing of semi-permanently submerged coastal forests (Rybczyk et al., 1998). Increased elevations would induce conditions well suited to seed germination and regeneration of cypress and tupelo trees (e.g. Day et al., 2012).

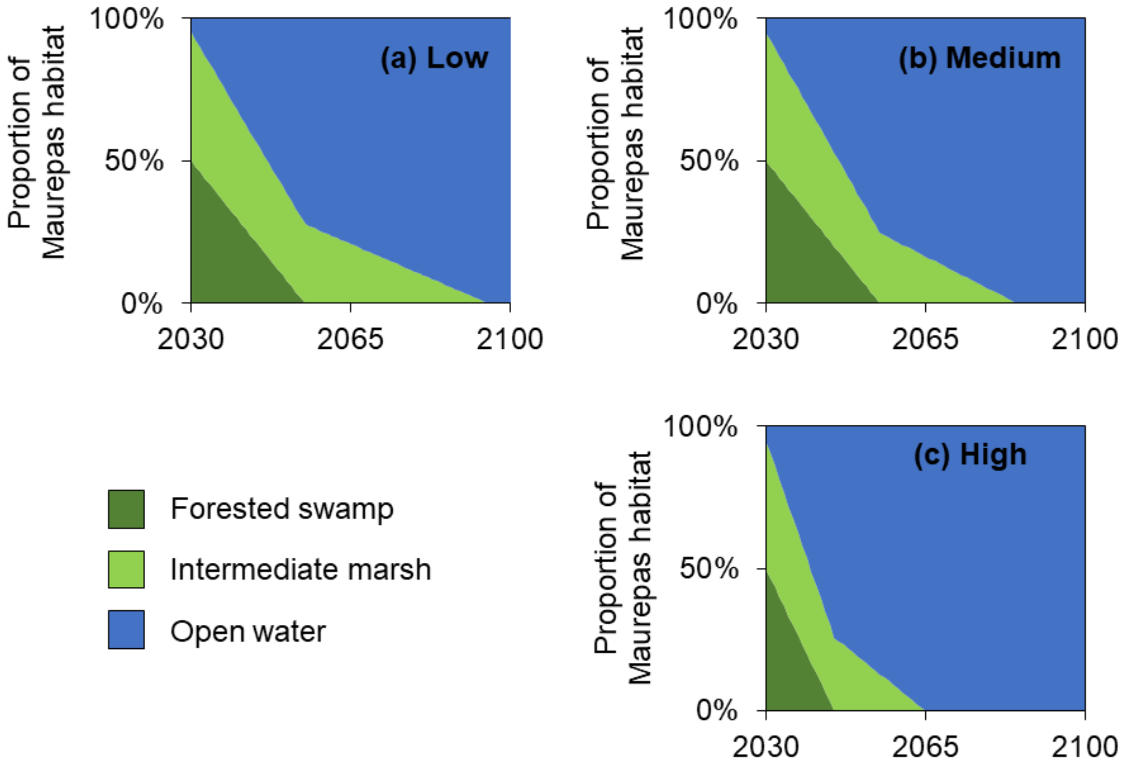


Figure 2: Proportion swamp, k_s, and marsh, k_m, in the Maurepas swamp under low, medium, and high SLR scenarios for 2030-2100 (based on CPRA, 2017b, and modified according to expert opinion – SI-B).

2.3 Ecosystem service valuation

We defined project benefits as the total value derived from provisioning, cultural, and regulating ecosystem services produced by the Maurepas swamp (using the framework of Hein et al., 2006). Ecosystem functions are typically converted to monetized ecosystem service flows (marginal and on a land area basis, e.g. $/acre/year) using market and non-market techniques (summarized in de Groot et al., 2002). However, conducting site-specific, individual valuation studies for ecosystem services would be time consuming and expensive. Appraisals, based on sales or pre-existing valuation studies are the norm for assessing housing or business values. Similarly, benefit transfer is applied for valuing natural systems. We specifically used a unit value transfer approach, where single value estimates from studies in the literature (termed "source studies"), deemed appropriate through a screening process, are "transferred" to the study site being valued (Rosenberger and Loomis, 2003). To carry out benefit transfer, we applied the protocol outlined in Rolfe et al. (2015) for applying benefit transfer with limited data. The literature search was conducted with the aid of the Ecosystem Valuation Toolkit (EVT), a tool developed by Earth Economics, which bins primary valuations by ecosystem service, habitat type, and location. Values from source studies were aggregated together to estimate the total ecosystem service flows in the Maurepas swamp (in 2014 $/acre/year).

Based on the source studies available in the EVT, this study values the Maurepas swamp for disaster risk reduction, water quality improvement, recreation, and carbon sequestration. Although we attempted to be as comprehensive as possible in valuing ecosystem services, no valuation can claim to be completely thorough. Given that all benefit transfer studies are limited by the quantity and quality of primary valuations available in the literature, the services we value only represent a subset of the numerous ecosystem services produced by the Maurepas swamp (see Batker et al., 2010). For example, cultural services which include aesthetic and educational benefits, are ill suited to monetary valuation techniques and are in general poorly represented in ESV studies (Chan et al., 2012). In addition, although the Maurepas swamp is not fished commercially, it does provide valuable nursery habitat for important commercial fish species like Gulf menhaden (Fox et al., 2007).

As we demonstrated in section 2.2, degradation of land and land restoration result in two types of environmental changes that must be accounted for in ecosystem service valuation. Quantitative changes, which describe the overall loss or gain of land area, can be valued by simply extrapolating monetized ecosystem service flows (based on the aggregate per acre values found in benefit transfer) over the total area affected by the diversion. Qualitative changes in condition describe the shifting nature of habitat type as the ecosystem degrades or is restored. Ideally source studies in the literature would cover the range of habitat types found. However, source studies are limited and this is not possible for all ecosystem services. To circumvent this issue, results of benefit transfer were assigned to the ecosystem service value for swamp habitat, and the marsh ecosystem service value was calculated by adjusting the swamp value proportionally according to a relevant biophysical parameter (SI-C.2).

Averaging values obtained from multiple source studies, the individual ecosystem service values for disaster risk reduction, water quality improvement, recreation, and carbon sequestration were $1,332.23, $283.20, $192.41, and $425.00 /acre/year (in 2014 US dollars, details in SI-C.2), respectively. Aggregating values, the ecosystem service value for swamp habitat, ESV_s, was calculated to be $2,232.83 /acre/year. Next, adjustments were made according to different biophysical parameters. For disaster risk reduction, ESV_s was adjusted proportionally according to water drag resistance (calculated at storm surge levels) and wind reduction parameters, accounting for differences in resistance to water motion and interaction between wind and water. For water quality improvement, the drag resistance parameter (calculated at average water level) was applied to account for differences in water residence time. For recreation, the same value (which considers recreational fishing and hunting only, but not other activities such as kayaking or bird watching) was applied for both swamp and marsh habitat, given that valuable recreation opportunities exist for both, and the difference cannot be teased out using biophysical data. For carbon sequestration, we refer to Mack et al. (2014) who reviewed the literature for sequestration rates in various habitats in Louisiana. Following these adjustments, the individual ecosystem service values for disaster risk reduction, water quality improvement, recreation, and carbon sequestration in marsh habitat were calculated to be $791.34, $529.58, $192.41, and $160.00

/acre/year (in 2014 US dollars, details in SI-C.2), respectively. Aggregating values, ESV_m was calculated to be $1673.33 /acre/year.

In all cases, the ecosystem service value for open water was assumed to be zero. Although open water does have value, it is increasingly abundant, with low, no, or possibly negative marginal value (for example, hurricanes gain power over open water), and significantly less than that of wetland habitat (e.g. Batker et al., 2010).

2.4 Cost-benefit analysis

The benefit of river diversion projects was defined using the benefit-cost ratio, *B:C*. Benefits were calculated as the net benefit of restoration, or the difference in ecosystem services provided by restored and unrestored land. Annual benefit of restored land was calculated as the product of total subaerial land built by the diversion, $A_{div}(t)$, and the ecosystem service value for swamp, ESV_s. Annual benefit of unrestored land was calculated based on the assumed FWOA trajectories for the Maurepas swamp (Figure 2), where the area of unrestored land affected by the diversion, $A_{div}(t)$, is proportioned into swamp and marsh (according to k_s and k_m), and transformed (with ESV_s and ESV_m) into ecosystem services. In calculating the ecosystem services of unrestored land, we assumed that the habitat proportions (k_s and k_m) in Figure 2 for any given year are homogeneous across the Maurepas swamp. *B:C* was then determined by integrating net project benefits over the total project lifespan (here 70 years) and dividing by the total project costs (equation 1). Diversion costs were based on a linear function calibrated to cost estimates and diversion size in the 2017 CMP (McMann et al., 2017a). The cost function was based on projects both selected and not selected moving forward to implementation (see SI-C.1). The discount rate, used to convert future benefits and costs to their present value, was set at 0%, based on the approach taken in the CMP (CPRA, 2012, 2017a).

$$B{:}C = \frac{\sum_{t=0}^{70}[A_{div}(t) * ESV_s - A_{div}(t) * (k_s(t) * ESV_s + k_m(t) * ESV_m)]}{\sum_{t=0}^{70} C(t)} \#(1)$$

2.5 Model scenarios

A number of tests were developed to compare infrequently operated (P>0) with continuous (P=0) diversions. First, we tested to find a relationship between land building and P and Q. Normalized land gain (in year 70) was plotted against P, holding Q fixed at 7079 m^3/s, and normalized land gain was plotted against Q, holding P fixed at 2 years. Next, we tested the potential for sustaining land gain in the Maurepas swamp with an infrequent diversion. For this, we modeled scenarios for various combinations of diversion size (Q = 1770, 3540, and 7079 m^3/s) and time inactive (P = 0-4 years) and the three SLR scenarios (low, medium, and high) over 70 years, until the end of the century (2100). Results report both total land gain and the year at which the delta begins to retrograde (retreat). Finally, we used the land building estimates and ecosystem service values to calculate *B:C* for the same diversion sizes, periods of time inactive, and SLR scenarios described for the analyses above. All simulations were assumed to begin in 2030 based on documentation from the 2017 CMP that estimates diversion projects require up to 6 years of engineering and design and up to 7 years of construction (McMann et al., 2017b).

Using the linear function calibrated to cost estimates in the 2017 CMP, averaged costs for 1770, 3540, and 7079 m^3/s sized diversions are: \$725, \$1,139, and \$1,966 million respectively (also in 2014 US dollars). Although the function provides a rough guide (and correlation between cost and discharge is strong, r^2 = 0.84), it should be noted that other factors are involved in determining cost, such as diversion structure and channel design.

3 Results

3.1 Diversion size and infrequent operation

Results demonstrated that land building is proportional to Q and inversely proportional to P (according to equations in Figure 3). This makes sense, given the assumption that the channel depth (and therefore the efficiency of sand capture from the river) remains the same between discharge levels, and also given that we neglected effects related to width and aspect ratio of the channel. This leaves sediment delivered to the delta as the only variable that varies with Q and P. For example, by doubling Q the volume of sediment delivered to the delta is doubled, and the results indicate that this approximately doubles subaerial land gain.

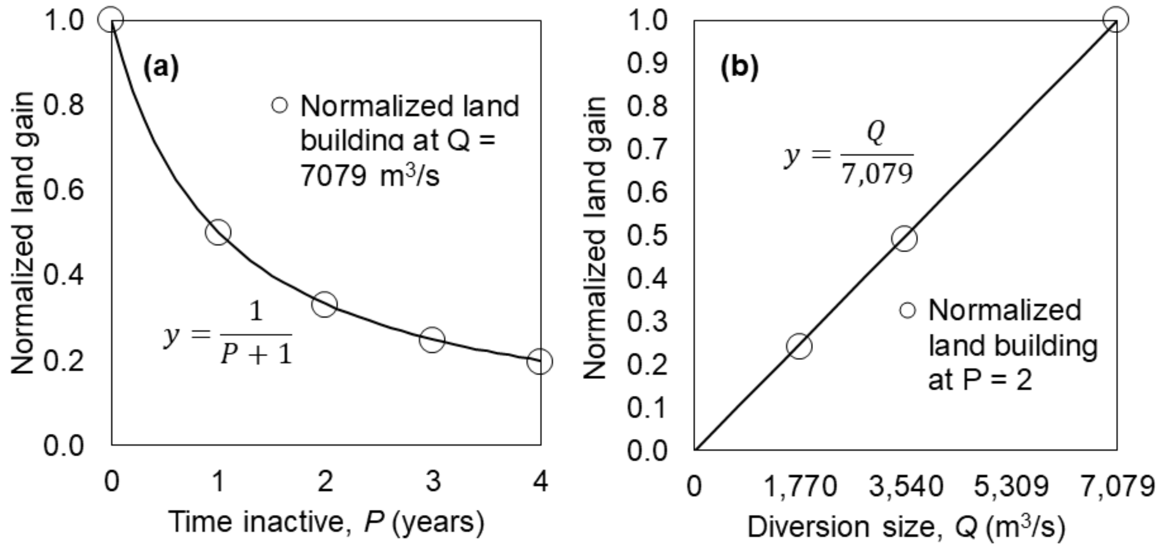


Figure 3: Normalized land building (against maximum land building achieved with Q = 7079 m^3/s and P = 0 years) related to (a) P, holding diversion size fixed at Q = 7079 m^3/s, and (b) Q, holding time inactive fixed at P = 2 years. Open circles represent modelling results and dark lines represent idealized relationships given by equations.

3.2 Potential for land building in the Maurepas Swamp

We tested how much land can be sustained in the Maurepas swamp and for how long. Under an accelerating SLR scenario, deltas formed using river diversions have a finite life span. In each modelled scenario, annual land gain had a diminishing trajectory, indicating that decline and submergence are ultimately inevitable with increasing sea level for the diversion sizes we modeled (Figure 4). Sustained land building (until the onset of retrogradation) for the next 50 years was possible in the low SLR scenario, 40 years in the medium SLR scenario, and 30 years in the high SLR scenario (Figure 5). No operation scenario modelled here allowed sustained land building beyond 100 years. Interestingly, the effect of intermittency and size on time until retrogradation was very minor – the dominant explaining factor was SLR. For example, in the medium SLR scenario the difference in duration of land building between the most aggressive (Q = 7079 m^3/s, P = 0) and least aggressive (Q = 1770 m^3/s, P = 4) diversion scenario is only 12 years. However, time inactive and size do have a significant effect on the amount of land that is built (Figure 6), just as described in the previous section.

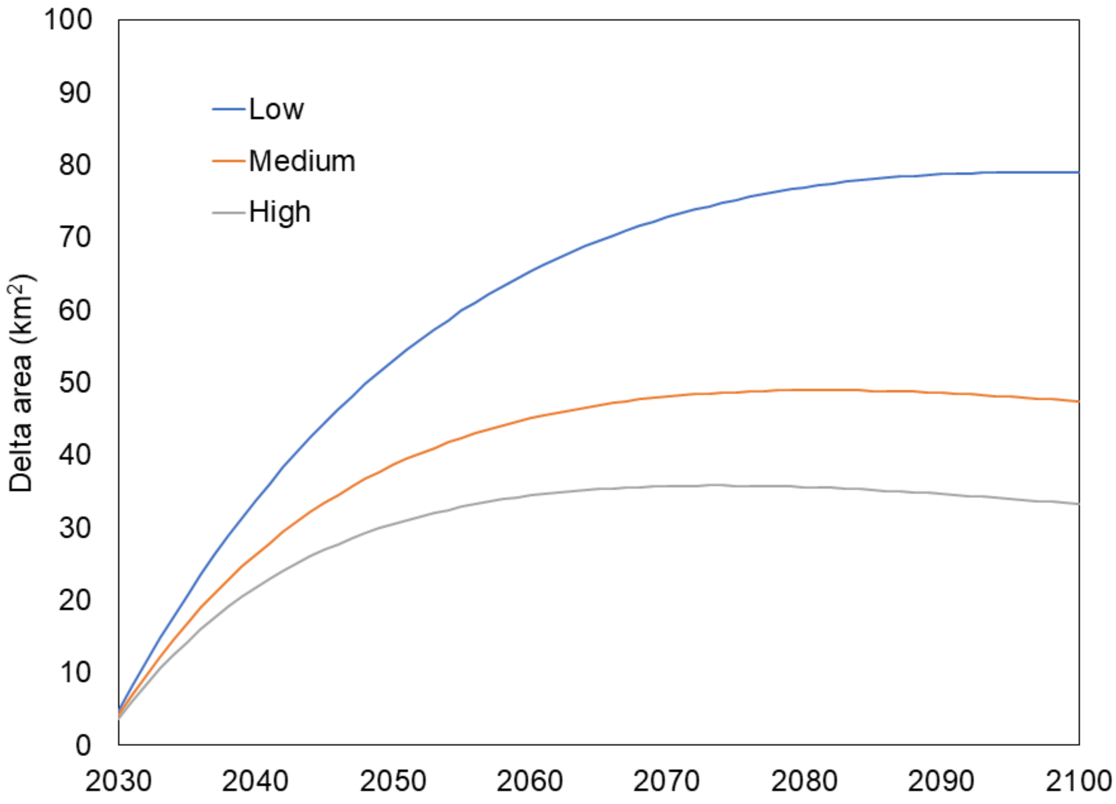


Figure 4: Land gain from 2030-2100 for a river diversion with Q = 7079 m^3/s and P = 2 years, demonstrating the diminishing trajectory that occurs with accelerating SLR.

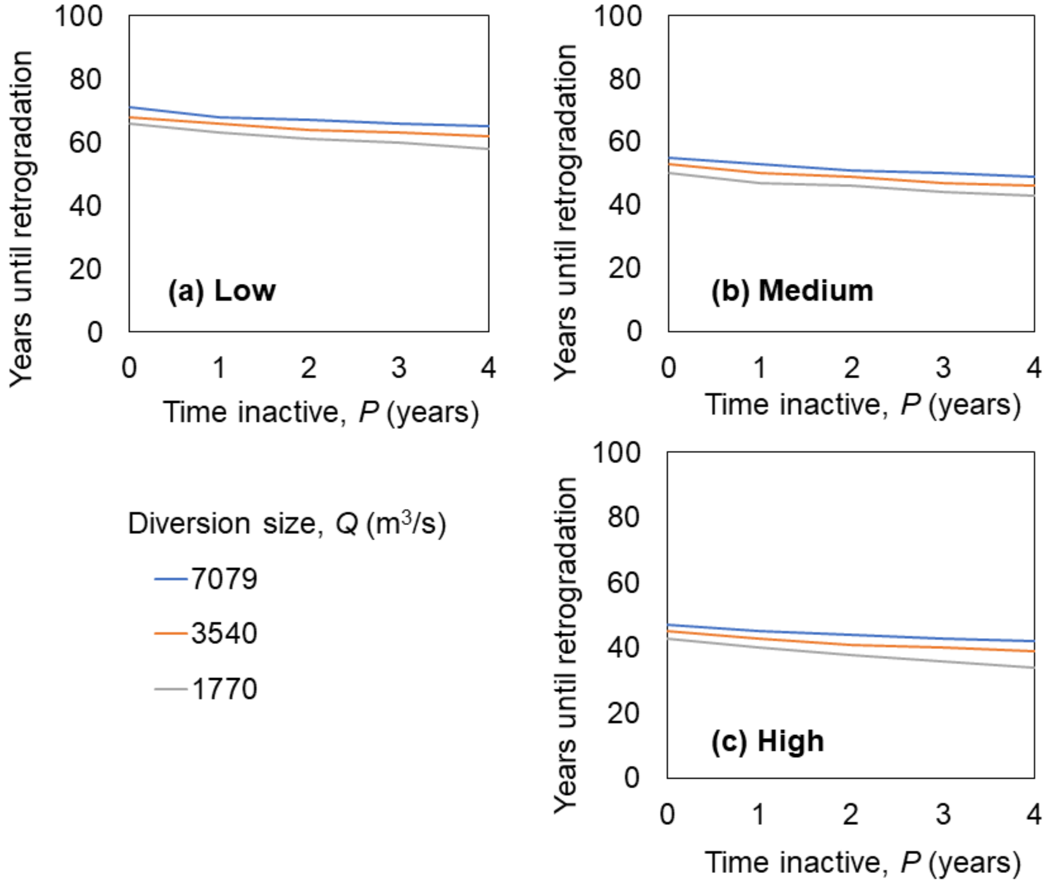


Figure 5: Duration of land building in the Maurepas swamp (given as years until retrogradation) versus time inactive between diversion operations, *P*, for three diversion sizes, *Q*, and three relative SLR scenarios: (a) low, (b) medium, and (c) high.

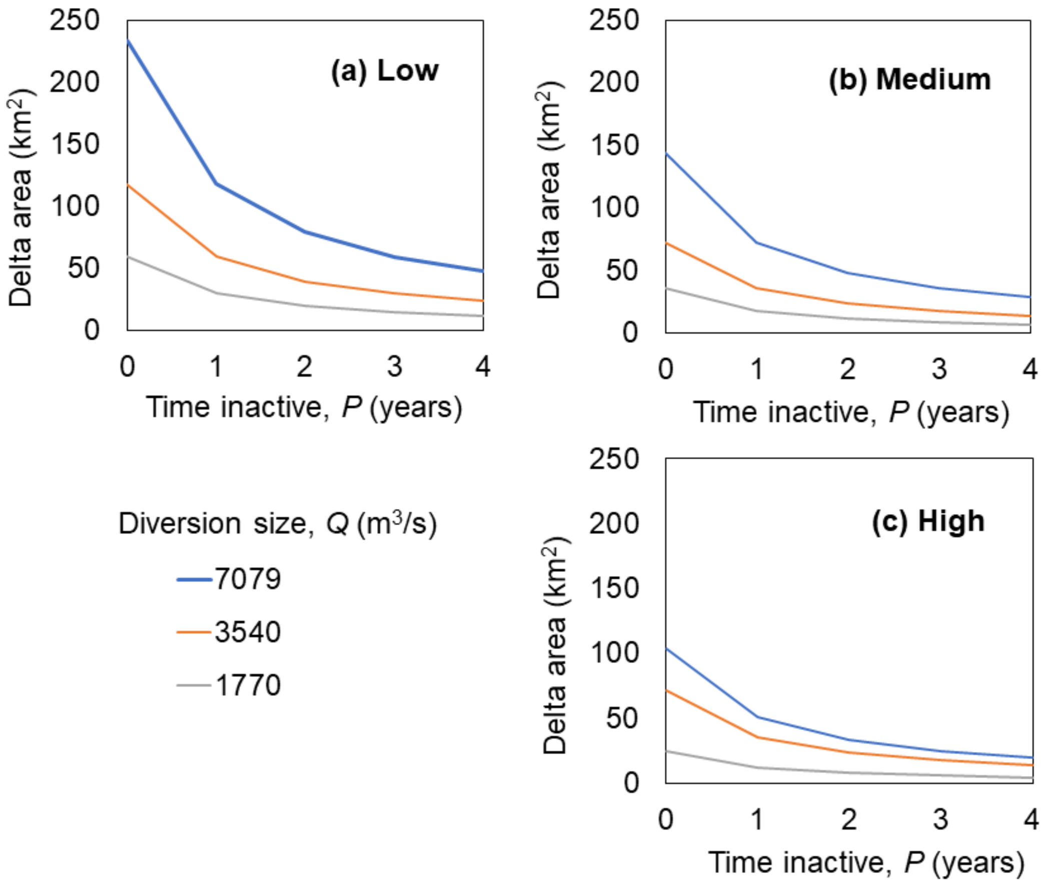


Figure 6: Amount of land built in the Maurepas swamp after 70 years versus time inactive between diversion operations, *P*, for three diversion sizes, *Q*, and three relative SLR scenarios: (a) low, (b) medium, and (c) high.

3.3 Benefit to cost ratio (B:C)

Considering a subset of ecosystem service benefits present at the site, and based on subaerial land gain, we calculated *B:C* for diversions of various *Q* and *P* into the Maurepas swamp (Figure 7). Notably, with the exception of the low SLR scenario, *B:C* is less than 1 for most infrequent diversion scenarios. A *B:C* <1 implies that the ecosystem service benefits we valued do not exceed project costs over 70 years for the immediate area of the delta splay. Our results for *B:C* do not demonstrate the same proportionality relationships (Figure 3) arrived at in section 3.1 because project cost increases with diversion size. *B:C* increases with greater diversion size, more frequent operation, and less severe SLR. It should be noted that these values for *B:C* are conservative since we did not value all ecosystem services provided by the Maurepas swamp; this is considered in more detail in the discussion.

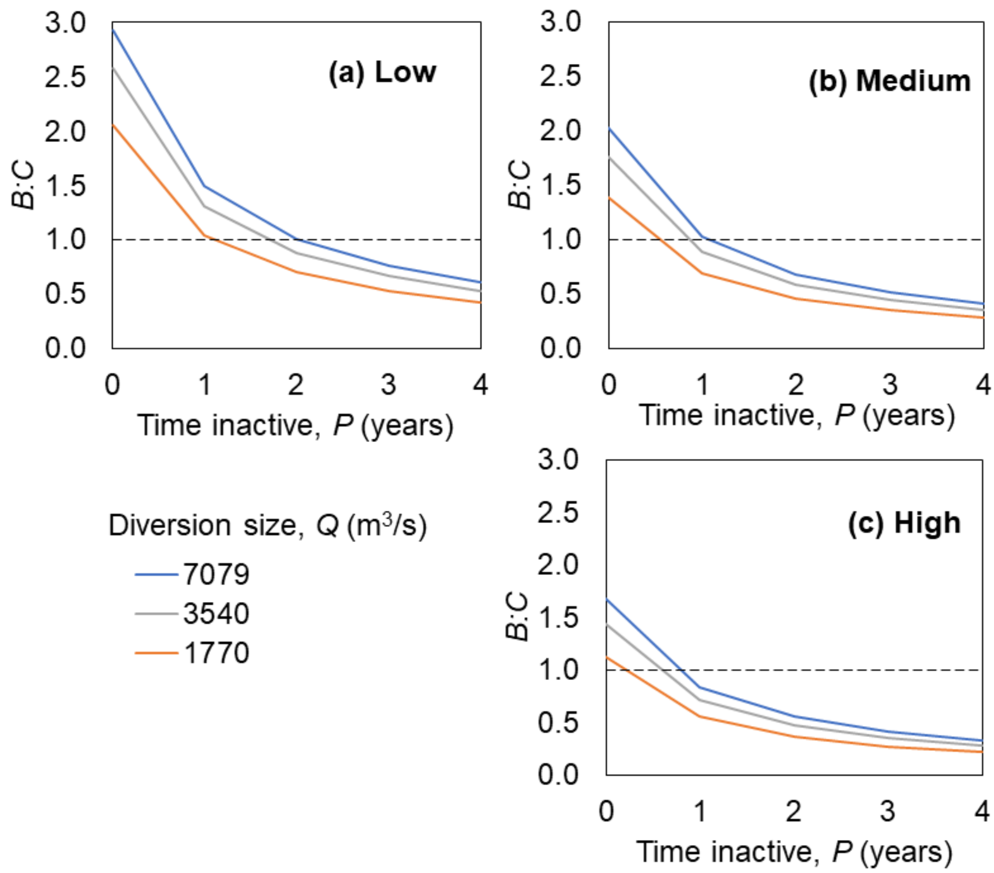


Figure 7: *B:C* versus time inactive between diversion operations, *P*, for three different sized diversions, *Q*, into the Maurepas swamp and three relative SLR scenarios: (a) low, (b) medium, and (c) high. The dotted line, where *B:C* = 1, is the value where the diversion becomes cost effective. These results are for benefits calculated with four ecosystem goods and services.

4 Discussion

4.1 Summary of findings and implications

Our results demonstrate a clear and well-defined physical tradeoff between land gain and P and Q (Figure 3). Defining operational period $P'=P+1$ (describing beginning of operation to the beginning of the next operation, including both active and inactive years), land gain with a diversion of Q and P' is equivalent to a diversion of $2Q$ and $2P'$. We also considered duration of land building, which we found is less related to Q and P, and is more controlled by severity of SLR (and also by subsidence, but which is less important in the Maurepas swamp). For example, two scenarios that provided about the same duration of land building (e.g. Q = 7079 m^3/s, P = 2, high SLR and Q = 1770 m^3/s, P = 2, high SLR) provide very different amounts of land building (33 km^2 and 8 km^2, respectively over 70 years). This occurred because, although the delta will initially grow quicker with higher sediment loads, over time more sediment must be used to just maintain the elevation of the existing delta, slowing the rate of progradation. Therefore, because maintenance requirements scale with discharge, it makes sense that both large and small diversions begin to retrograde at the same time (in the same SLR scenario). The SLR scenario, which describes the rate of increase in water level in the receiving basin, dictates the amount of sediment needed to maintain elevation of the existing delta. Therefore, many of the infrequent diversion scenarios, while supporting less land gain compared to a continuous diversion of the same size, still allowed comparable durations of land building, even in the high SLR scenario. It should be noted that all diversion sizes used in this study can be considered large, since the largest diversion proposed in the CMP is about 2124 m^3/s (McMann et al., 2017a). In addition, the Union Freshwater diversion proposed for the Maurepas swamp is only 708 m^3/s, smaller than the diversions modeled here.

Converting from land gain to benefits with ecosystem service values, ESV_s and ESV_m, allowed the comparison of benefits to costs with a common unit (dollars). By including consideration of diversion costs, we identified another tradeoff between land gain and *Q* and *P*. With the exception of the low SLR scenario, curtailing a diversion for a year or more (P>0) reduced *B:C* < 1 for the four EGS considered (Figure 7). Considering costs, a diversion with 2*Q* and 2*P'* is less preferable to a diversion of *Q* and *P'*, because diversion cost increases with *Q*. However, our values for *B:C* are conservative since we did not include all benefits in our analysis (more below).

4.2 Secondary effects of diversion operation

Based on our results, if land gain (and the related ecosystem service benefits) are our objective, then there are economic disadvantages to operating a diversion less than its maximum allowable duration (and consequently economic disadvantages to using a large intermittent diversion of 2*Q* and 2*P'* instead of a smaller continuous diversion of *Q* and *P'*). However, there are other effects of diversions not included in our model that should be considered. These effects, both within and outside the zone of land building, include water quality (e.g. salinity and nutrients), inundation, fisheries, and management of the MR.

Diversions (both sediment and freshwater) have basin-wide effects on water quality (Wang et al., 2017). Large inputs of freshwater significantly reduce salinity (Das et al., 2012; Wang et al., 2017), sometimes enough to cause significant marine community shifts (depending on the salinity tolerance of species). Although the overall effect of freshwater input is an increase in estuarine primary and secondary productivity (Viosca, 1938; Gunter, 1953; Day et al., 1989; Guillory, 1999) and modelling efforts have shown mostly increases in fisheries catch with large CMP diversions (REC & EE, 2016), the displacement of marine organisms from diversions has provoked conflict between commercial fishermen and proponents of diversion projects. It is possible that infrequent operation would alleviate concerns related to over-freshening, given that salinity would return to normal shortly after an opening (Lane et al., 2004). Nutrient laden water from diversions can also cause rapid phytoplankton growth, potentially resulting in harmful algal blooms. Past blooms resulting from Bonnet Carre spillway openings were short lived and not dominated by harmful algal species (Turner et al., 2004; White et al., 2009; Bargu et al., 2011), but in a few cases these algal species were replaced by toxin producing cyanobacteria (Turner et al., 2004; Bargu et al., 2011). Evidence suggests that harmful cyanobacteria are common in the wider coastal zone and operating a large diversion infrequently may serve to temporarily flush the cyanobacteria from the system (Day et al., 2009; Riekenberg et al., 2015; Roy et al., 2016). Therefore, infrequent diversions might support greater levels of ecosystem services compared to continuous diversions by reducing deleterious effects related to over-freshening and harmful algal blooms, but these linkages must be tested in a more comprehensive modelling effort.

The effects of water quality and inundation on wetlands themselves must also be considered. In this study's ecosystem service valuation, we considered the ability of wetlands to improve water quality through denitrification (e.g. processing of nutrients in wastewater or diverted river water). However, it is debated whether excessive nutrients lead to the decomposition of below ground biomass and the loss of soil strength, increasing the susceptibility of wetlands to hurricane damage (Darby and Turner, 2008; Turner et al., 2009; Kearney et al., 2011; Day et al., 2013; Graham and Mendelssohn, 2014; Nyman, 2014). In addition to nutrients, diversions will cause greater inundation in wetlands which, with increasing duration, results in lower biomass production in marshes (Snedden et al., 2015) and swamps (Pezeshki and Anderson, 1996). For example, baldcypress trees require a complete drawdown of water to near dry conditions for several months for seeds to germinate and establish, and then seedlings must grow tall enough to stay above the next flood event (Conner et al., 1986). Ultimately, added nutrients and flooding from diversions will affect the ecosystem functioning in some way, potentially resulting in ecosystem service losses. Although infrequent operation might reduce such losses, these effects haven't been quantified in our model.

Considering the 2017 CMP, there will likely be a suite of diversions (both sediment and freshwater) operating along the river in the future. However, the water and sediment available in the MR is finite, and several large diversions operated simultaneously would lower the river stage to such a level that it would begin to affect activities such as shipping (for example, a 7079 m^3/s river diversion is already 34% of an average 4-month spring flow – 20,624 m^3/s, March 1-June 30 – of the MR). Infrequent operation presents an opportunity to stagger diversions among separate years. In addition, river diversions also provide strategic benefits when the river stage is high. Diversions, such as the Bonnet Carre spillway, are a means of lowering the river stage, thus reducing pressure on river levees and lowering the risk of flooding to urban areas.

The secondary (in addition to land gain) effects of diversions discussed here – water quality, flooding, and river management – will affect ecosystem services within and beyond the zone of land building, and consequently will affect *B:C*. Although the literature suggests evidence that infrequent operation (increase in *P*) could increase *B:C*, a formal systematic analysis is required. Basin wide effects and the resulting benefits (and costs) of both continuously and infrequently operated diversions must be incorporated into this type of BCA using hydrodynamic modelling capable of simulating sediment transport, nutrient dynamics, and salinity (e.g. the CPRA's Integrated Compartment Model, Brown et al., 2017).

4.3 Limitations of study and opportunities for future research

Benefit transfer, the technique used in this study to calculate ecosystem service benefits, is popular amongst practitioners who do not possess the time or resources to conduct an original valuation. Benefit transfer involves "transferring" values from their original location and context (Plummer, 2009) and, although we took care to follow recommended methods (SI-C), our estimates are still subject to error. For example, our study resulted in a value of $2,233 /acre/year for swamp, with an uncertainty of +116% and -79%, and other studies have calculated values for forested wetland near $10,000 /acre/year (Batker et al., 2010; Costanza et al., 2014). The difference between our value and those of other studies is due to a number of reasons. Based on the source studies available in the EVT, and our chosen filtering criteria (SI-C), not all ecosystem services provisioned by the Maurepas swamp were included. It is in general the case that some ecosystem services, such as carbon sequestration, are well suited for valuation (since benefits are non-proximal, the social cost of carbon can usually be applied). Whereas other cultural services, such as education and aesthetic values, are less so and source studies are not readily available (Chan et al., 2012; Luisetti et al., 2014). Additionally, this is a very local study so only source studies from states bordering the Gulf of Mexico were accepted, where on the other hand Batker et al. (2010) and Costanza et al. (2014) chose broader boundaries (nationwide and global, respectively). The question of whether to accept broader boundaries, or accept that some ecosystem services are not included, leads to limitations in the results. In our case, the values of the particular EGS used in the study are probably closer to the true values for the Maurepas swamp, since they originate from a closer proximity, but our total aggregate value is still an underestimate since we only considered a subset of services. Also, this study chose to apply a "final ecosystem goods and services" framework (Boyd and Banzhaf, 2007; Fisher et al., 2009; Landers and Nahlik, 2013), where supporting services are not included (see SI-C for more detail). Batker et al. (2010) and Costanza et al. (1997, 2014) included supporting services in their valuations. Finally, our study and those of Batker et al. (2010) and Costanza et al. (1997, 2014) are conducted at different geographic scales, so the populations benefitting from the ecosystem services are different (e.g. the Maurepas swamp doesn't provide a water supply to any of the surrounding communities). These issues highlight the uncertainty in using benefit transfer for valuation and the differences that result from various methodological choices. These issues will hopefully be resolved as researchers develop spatial models with closer linkages between ecosystem service flows and the people who benefit (Bagstad et al., 2014; Villa et al., 2014). These models are in the early stages of development, but should be incorporated into BCA like this study as soon as they are available.

5 Conclusions

From a broader perspective, this study developed a tool which could be applied to any river diversion project in a coastal setting. This is relevant globally; given deltas are both among the most valuable and the most stressed systems in the world (Giosan et al., 2014; Vorosmarty et al., 2009; Syvitski et al., 2009), it is important to develop tools that support their "sustainable stewardship" (Scharin et al., 2016).

The specific application of this study was in evaluating tradeoffs in implementing a large, infrequent diversion into the Maurepas swamp. To accomplish this, we evaluated several scenarios for diversions of various sizes and times inactive under three trajectories for relative SLR. Infrequent (P>0) diversions allowed comparable durations of land building compared to continuous (P=0) diversions, and the amount of land built was significant (e.g. 51 km^2 for a 7079 m^3/s diversion with P = 1), even under high SLR. Using our BCA framework, with the exception of the low SLR scenarios, *B:C* was less than 1 for most of the large, infrequent diversion scenarios. Compared to smaller diversions operated more often, with our framework there are economic disadvantages to large, infrequent diversions given that costs positively correlate with *Q*. However, our benefits only included a subset of ecosystem services provided through subaerial land gain, and did not consider the secondary effects (positive or negative) of water quality, flooding, and MR management. Data from the literature suggests that infrequent operation would provide additional ecosystem service benefits to the broader Lake Maurepas-Pontchartrain-Borgne ecosystem by minimizing long-term changes in salinity and water quality, reducing inundation time, and allowing for greater consolidation of soils between diversion pulses. Such linkages were not developed in this paper, therefore future work should incorporate water quality modelling into this BCA framework to further assess the *B:C* of large, infrequent diversions.

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