Efecto de la urbanización y el cambio climático en los ecosistemas de playas arenosas

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Beyond a critical point within a finite space, freedom diminishes as numbers increase. This is as true of humans in the finite space of a planetary ecosystem as it is of gas molecules in a sealed flask. The human question is not how many can possibly survive within the system, but what kind of existence is possible for those who do survive.

Frank Herbert. Dune.

Más allá de un punto crítico dentro de un espacio finito, la libertad disminuye a medida que aumenta el número. Esto es tan cierto para los humanos en el espacio finito de un ecosistema planetario como para las moléculas de gas en un matraz sellado. La cuestión humana no es cuántos pueden sobrevivir dentro del sistema, sino qué tipo de existencia es posible para aquellos que sobreviven.

Frank Herbert. Dune.

RESUMEN

Las playas arenosas abarcan dos tercios de las costas libres de hielo del mundo. Estos ecosistemas de interfase entre el mar y la tierra, considerados entre los más dinámicos del planeta, están asociados con importantes funciones ecológicas. La abundancia de alimento y la provisión de refugio contra depredadores los transforma en hábitats clave, permanentes o transitorios, para la reproducción, migración, alimentación y cría de variadas especies de mamíferos, aves, peces e invertebrados. Además, brindan servicios ecosistémicos de provisión de alimento (peces y bivalvos), protección contra la acción erosiva del mar, recreación, desarrollo cognitivo y mejora en la calidad de aire y agua, entre otros. En especial, las playas arenosas constituyen un pilar de la producción turística, lo que las asocia aún más al desarrollo económico y urbano. La fuerte preferencia humana por establecerse en áreas costeras aumenta la dependencia de esta zona para alojamiento, infraestructura, comercio, industria y recreación. Como resultado de este patrón demográfico se producen presiones adversas, a las cuales se agregan las repercusiones del cambio climático. Incorporar la urbanización, la actividad humana y el cambio climático al marco ecológico es de suma relevancia en el contexto del Antropoceno. En este sentido, las playas arenosas constituyen un interesante modelo de estudio, pues se ven afectadas directamente por una gran exposición a la urbanización y por los efectos del cambio climático, los cuales pueden integrarse en un extenso marco teórico ecológico. En esta tesis se aborda dicha temática mediante aproximaciones diversas que incluyen la estimación de variables críticas en base a información satelital. En primera instancia, se aborda el efecto del clima sobre el área de playas mediante el análisis de imágenes satelitales y algoritmos de clasificación semi-automática. Los resultados mostraron una correlación significativa entre el área de las playas en la costa de Montevideo y variaciones en las condiciones climáticas locales y globales, destacándose el nivel del mar, las condiciones de viento y la temperatura media del año previo, así como una asociación a los eventos La Niña/ El Niño. El análisis a escala multidecadal mostró un ciclo de 27 años con fases de acreción y erosión bien delimitadas y relacionadas con la configuración climática en la región. El modelo derivado de este análisis permitirá realizar estudios de predicción de los efectos del cambio climático para esta costa. El hecho de haber empleado información de acceso libre y de haber detallado la metodología, facilitará la replicación de análisis de este tipo. En segunda instancia, se analiza la importancia de la urbanización, estimada mediante la densidad poblacional y las luces nocturnas, como descriptor de la riqueza de especies en la playa. Se encontró una correlación negativa entre el grado de urbanización y la riqueza de especies macrobentónicas de playas arenosas de la costa uruguaya afectada por el gradiente estuarino del Río de la Plata. La salinidad aparece como el principal forzante de la riqueza macrobentónica para el área: playas con salinidad ≥27.2 mostraron las mayores riquezas de especies, mientras que aquellas con menos salinidad y mayores luces nocturnas mostraron la menor riqueza específica. No obstante, la urbanización afecta la severidad del ambiente y las interacciones bióticas, por lo que debe incluirse en estudios de ecología de playas en el Antropoceno. Por último, se analizan las implicaciones de las características urbanas, ecosistémicas y los distintos patrones de uso, en el manejo de ecosistemas arenosos. Se incorporaron indicadores cuantitativos disponibles a nivel mundial a efectos de modelar el potencial de recreación y conservación, y los resultados se contrastaron con las percepciones de los expertos. Las luces nocturnas y la magnitud del turismo estacional, ambas variables antropogénicas, mostraron el mejor desempeño en la explicación de la opinión de los expertos. Incorporar variables cuantitativas posibilita un análisis más formal, reduciendo

la subjetividad derivada de la percepción. Un enfoque que combine la información científica derivada de variables cuantitativas mediante información satelital y el conocimiento de expertos, podría conducir a una aproximación más objetiva sobre el uso potencial de las playas. En conjunto, esta tesis presenta un abordaje más formal que los actualmente disponibles para evaluar los efectos de la urbanización y el cambio climático en los ecosistemas de playas, con resultados que pueden ser relevantes en el análisis y manejo de estos ecosistemas en el contexto del Antropoceno. Se destaca la necesidad de la inversión en investigación y su incorporación a los sistemas de gestión para afrontar la convergencia de una mayor presión humana y la degradación del ambiente, en un contexto climático adverso.

Palabras clave: Antropoceno, Ecología, Ecosistemas de playas, Información satelital, Urbanización.

ABSTRACT

Sandy beaches make up two-thirds of the world's ice-free coasts. These interface ecosystems, located between the sea and the land, are considered among the most dynamic physical systems on the planet and are associated with important ecological functions. The abundance of food, and the provision of shelter against predators transform them in critical habitats, permanent or transitory, for the reproduction, migration, feeding and breeding of several species of mammals, birds, fishes and invertebrates. In addition, these ecosystems provide services from which the society obtains goods and benefits, including food (fish and bivalves), protection against the erosive action of the sea, recreation, cognitive development and improvement of air and water quality, among others. Sandy beaches constitute a pillar of tourism production, which is associated with economic and urban development. The strong human preference for settling in coastal areas increases the dependence on the coastal zone for accommodation, infrastructure, commerce, industry and recreation, leading to adverse environmental pressures that act together with climate change stressors. Incorporating urbanization and human activity into the theoretical ecological framework is highly relevant in the context of the Anthropocene. In this sense, sandy beaches constitute an interesting study model, as they are directly affected by a great exposure to urbanization and the effects of climate change, which can be integrated into an extensive theoretical ecological framework. In this thesis, this issue is addressed by several approaches that include the use of quantified variables of ecological importance based on satellite information. First, the effects of the climate on beach area are addressed through the analysis of satellite images with semi-automatic classification algorithms. The results showed a significant correlation between the area of the beaches on the coast of Montevideo and variations in local and global climatic conditions, including sea level, wind conditions and temperature in the previous year, as well as an association with the La Niña/ El Niño events. Long-term trends described a 27-year cycle with well-delimited quasi-decadal erosion and accretion phases related to climatic configurations. The model derived from this analysis can be used to predict the effects of climate change on this coast. The use of open-access information and the detailed description of the methodology foster this kind of analysis. Second, human population density and nighttime lights were assessed as indicators of urban impact on sandy beach macrobenthic species richness in the Uruguayan coast, along the strong environmental gradient defined by the Rio de la Plata. Salinity was the dominant ecological driver: beaches with salinity ≥27.2 showed higher species richness, while beaches with less salinity and higher nighttime lights showed the lowest species richness. These results highlight the relevance of including urbanization-related variables in sandy beach ecological studies in the Anthropocene. Finally, the implications of urban and ecosystem characteristics and the different patterns of use on the management of these sandy ecosystems are analyzed. Quantitative indicators available worldwide were incorporated to model recreation and conservation potential, and the results were contrasted with expert perceptions. Nighttime lights and the magnitude of seasonal tourism, both of anthropogenic origin, showed the best performance as predictors of expert opinion. Incorporating quantified variables enables a more formal analysis, reducing the subjectivity inherent to perceptions. A combined and complementary approach merging scientific information derived from satellite-based information and expert knowledge could lead to a coherent management narrative about the potential use of beaches for recreation and/or conservation. As a whole, this thesis presents a formal approach to assess the effects of urbanization and climate change on beach ecosystems, with results

that can be relevant in the analysis and management of these ecosystems in the Anthropocene. There is a short-term need for investment in research, and its incorporation into management systems, to face the synergistic effects of increasing human pressure and environmental degradation, under an adverse climate.

Keywords: Anthropocene, Beach ecosystems, Ecology, Satellite information Urbanization.

TABLA DE CONTENIDO

1. INTRODUCCIÓN GENERAL

1.1 Ecología y Antropoceno

Desde los inicios de la Ecología como ciencia ha estado claro que su foco de estudio no debe estar limitado a los “sistemas naturales”, sino que debe integrar todo tipo de sistemas ambientales, trascendiendo el origen de los mismos (Tansley 1935). Incorporar la mayor variedad de ambientes posibles tiene un valor intrínseco, ya que la heterogeneidad de los sistemas de estudio genera diferentes aproximaciones que han dado lugar a una historia de debates que enriquecen el marco conceptual ecológico (Deléage 1993). Debido al impacto significativo de las actividades humanas sobre la geología y ecología del planeta, se ha denominado a la época histórica actual como Antropoceno (Crutzen & Stoermer 2000). Este impacto no se limita al cambio climático, sino que existe amplia evidencia de la creciente influencia humana sobre gran cantidad de variables ambientales (Ruddiman 2013). La mayoría de estas tendencias comienzan en los últimos 150 años, pero en especial experimentan una fuerte aceleración a partir de la industrialización (Crutzen & Stoermer 2000, Steffen *et al.* 2015). Asociado al crecimiento industrial se ha suscitado un crecimiento acelerado de la población humana y una concentración en urbes (Grimm *et al.* 2008, Steffen *et al.* 2015). En 1900 apenas un 10% de la población global era urbana, hoy en día este porcentaje excede el 50% y se espera que llegue al 60% (4.9 mil millones) para el 2030 (Fig. 1).

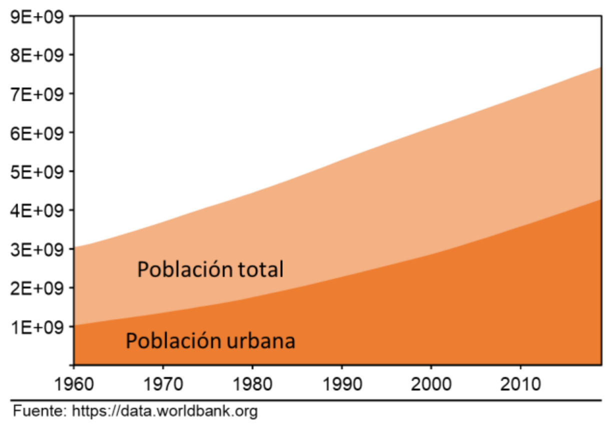

Figura 1. Evolución de la población humana total del planeta y la población urbana para el período 1960 – 2019. Datos del Banco Mundial (https://data.worldbank.org/).

Al ambiente urbano se le adjudican el 78% de las emisiones globales de carbono, el 60% del uso de agua y el 76% del uso industrial de madera (Grimm *et al.* 2008). La alteración de los sistemas hidrológicos (Bodini *et al.* 2012) junto con la emisión de gases de efecto invernadero (Ruddiman 2013), hacen que el rol primario de las ciudades en el cambio climático esté más allá de cualquier duda (Grimm *et al.* 2008, Garschagen & Romero-Lankao 2013, Revi *et al.* 2014, UN 2014). Sumado a los efectos globales, la urbanización tiene efectos negativos a escala local, afectando la temperatura, la calidad del agua, los ciclos de carbono y la química atmosférica (Gaston 2010). De hecho, muchos de los problemas ambientales presentes en áreas urbanas son generados localmente, ya que procesos como impermeabilización de superficies (pavimentado), concentración de habitantes, canalización de los flujos de agua, tráfico, etc., afectan la estructura y el funcionamiento de los ecosistemas (Wu 2014).

Durante el Siglo XX, la Ecología incorporó progresivamente a las ciudades como ambiente de estudio. Sin embargo, como ha ocurrido históricamente en el desarrollo de esta ciencia, aquellos ecosistemas asociados a latitudes altas o medias de Europa y los Estados Unidos están sobrerrepresentados en la literatura, mientras que los ecosistemas asociados a latitudes bajas son analizados en menor medida (Bellocq *et al.* 2017). En el caso específico de las playas urbanas, los estudios son escasos y en general se limitan a comparar ecosistemas alterados y prístinos (Machado *et al.* 2016) o dependen de índices de percepción para estimar la urbanización (González *et al.* 2014). Ambas aproximaciones limitan la rigurosidad de los análisis y la capacidad de realizar comparaciones de macroescala. Esta tendencia es particularmente llamativa si se considera que 17 de las 20 ciudades más grandes del mundo son costeras (UN 2014).

La fuerte preferencia humana por establecerse en áreas costeras (Small & Nicholls, 2003) aumenta la dependencia de esta zona para alojamiento, infraestructura, comercio, industria, turismo y recreación (Defeo *et al.*, 2009; Sengupta *et al.*, 2020). Como resultado de este patrón demográfico se producen presiones adversas, a las cuales se agregan las repercusiones del cambio climático, conformándose el denominado *"triple whammy"*[1] (Defeo & Elliott 2020), que está dado por:

(1) aumento de la urbanización y la industrialización

(2) aumento en el uso de recursos, como agua, espacio y alimentos provenientes del mar

[1] '*whammy*' se define como algo que tiene un gran impacto, generalmente negativo, coloquialmente un golpe de puño (Collins English Dictionary)

(3) aumento de la susceptibilidad y disminución de la resiliencia y resistencia de los ecosistemas a los efectos del cambio climático (e.g. calentamiento del agua).

Las playas arenosas se encuentran entre los ecosistemas más frecuentes dentro de la zona costera (McLachlan & Defeo 2018), constituyendo un pilar de la producción turística, lo que las asocia al desarrollo económico y urbano, así como fuente de recursos materiales (arena) y producción de alimentos (peces y bivalvos). El contexto de creciente urbanización y marcada preferencia humana por zonas costeras, ejerce una presión sobre los servicios ecosistémicos generados por las playas. Para conservar dichos servicios se requiere un conocimiento ecológico detallado de: 1) el funcionamiento de estos sistemas y 2) sus relaciones con la urbanización (McLachlan & Defeo 2018). Esta tesis se enfoca principalmente en el segundo punto, en el entendido de que existe un gran potencial de avance a través de la incorporación sistematizada de la urbanización a la ecología de playas, aprovechando y enriqueciendo el desarrollo teórico existente.

1.2 Las playas como ecosistemas

Las playas arenosas son ambientes dinámicos que abarcan dos tercios de las costas libres de hielo del mundo (McLachlan & Defeo 2018). Estos ecosistemas de interfase entre el mar y la tierra son considerados entre los sistemas físicos más dinámicos del planeta (Short 1999). Las playas arenosas y su faja dunar están asociadas con importantes funciones ecológicas como alta producción primaria y secundaria (Odebrecht *et al.* 2009) y ciclado de nutrientes (Constanza *et al.* 1997). La abundancia de alimento, sumada a la provisión de refugio contra depredadores, las transforma en hábitats clave, permanentes o transitorios, para la reproducción, migración, alimentación y cría de variadas especies de mamíferos, aves, insectos y peces (Selleslagh *et al.* 2012, Caballero 2014, Mourglia *et al.* 2015). Además, brindan servicios ecosistémicos de provisión de alimento (peces y bivalvos), protección contra la acción erosiva del mar, recreación, desarrollo cognitivo y mejora en la calidad de aire y agua entre otros (Defeo *et al.* 2009, Jackson & Nordstrom 2011).

Las playas arenosas proveen hábitat para una diversidad de organismos, y el foco tradicional de la ecología de estos ecosistemas se ha centrado principalmente en las comunidades macrobentónicas, compuestas por especies de invertebrados altamente adaptadas a este ambiente extremo (Lercari & Defeo 2006). La mayoría de estas especies no se encuentran en otros ecosistemas, sus adaptaciones para la vida en las playas abarcan: movilidad, enterramiento, exoesqueletos protectores y mecanismos de orientación, entre otras (Defeo *et al*. 2009). Esta fauna incluye representantes de varios *phyla*, siendo usualmente dominantes los crustáceos, moluscos y poliquetos, que cumplen roles variados en la red trófica, incluyendo depredadores, carroñeros y filtradores (Lercari *et al.* 2018). Esta comunidad incorpora a la red trófica materia orgánica de origen terrestre y acuático, altamente relevante para estos ecosistemas (Bergamino *et al*. 2013). Dado su endemismo, adaptación al medio y rol trófico la macrofauna constituye un indicador ecosistémico extensamente utilizado en playas arenosas. Altas diversidades de macrofauna se asocian con mayor biomasa en niveles tróficos superiores (Lercari *et al.* 2010) y con mayor resiliencia frente a efectos del cambio climático (Lercari *et al.* 2018), así como con las características del ambiente físico (Barboza & Defeo 2015).

Las características únicas de los ecosistemas de playas las hacen útiles como modelos para el desarrollo de enfoques teóricos y prácticos (Defeo *et al.* 2003, Defeo & McLachlan 2005). La hipótesis autoecológica sostiene que, en ambientes controlados físicamente, las comunidades están estructuradas por las respuestas de las especies individuales al ambiente físico, por encima de las interacciones entre especies (Noy-Meir 1979). La macrofauna es controlada mayormente por factores físicos como el tamaño de grano, la energía del oleaje, las mareas y la pendiente (McLachlan & Defeo 2018). El tamaño de grano en la playa está relacionado con el perfil de rompiente de las olas: olas de gran energía que rompen bruscamente depositan granos gruesos, mientras que las olas que rompen gradualmente disipan su energía y depositan granos finos. El tamaño de grano se relaciona con el ancho de swash a través del nivel de percolación del agua de la ola en la arena, y de esta manera los granos gruesos permiten una mayor percolación, reduciendo el área de swash en el caso de playas reflectivas. Además, el grano de menor tamaño requiere menos energía para su transporte por el viento, favoreciendo una deposición más uniforme, lo cual se refleja en una pendiente suave (Short 1999), que condiciona el patrón de rompiente de las olas. La interacción entre estas variables físicas genera un continuo de estados morfodinámicos cuyos extremos están definidos por playas disipativas (pendiente suave, grano fino y disipación gradual de la energía del olaje) y reflectivas (pendiente pronunciada, grano grueso y disipación brusca de la energía del olaje). En relación con la hipótesis autoecológica, la hipótesis de exclusión del swash (Swash Exclusion Hypothesis) sostiene que el clima de swash condiciona en mayor grado la presencia de la fauna: comunidades macrobentónicas más diversas y abundantes han sido correlacionadas con playas disipativas, las cuales son definidas como ecosistemas de menor estrés físico (Defeo & McLachlan 2013). La hipótesis de severidad de hábitat (Hábitat Harshness Hypothesis) amplía el espectro de predicciones de la hipótesis anterior al nivel poblacional, incluyendo potenciales relaciones inter e intraespecíficas (Defeo et al. 2003).

Existen otras variables físicas relevantes en la explicación de patrones faunísticos en playas arenosas. En este contexto, a escala macroecológica existe un incremento de la diversidad con la temperatura del agua (Barboza & Defeo 2015), reforzando el patrón altas diversidades asociadas a altas tasas metabólicas o limitaciones térmicas reducidas en organismos marinos (Tittensor *et al.* 2010, Defeo *et al*. 2017). El incremento en el rango de marea aumenta la diversidad, ensancha la playa y varía la ubicación de las zonas hidrodinámicas a lo largo del intermareal, reduciendo el tiempo de exposición al oleaje en una zona específica y aumentando la disponibilidad de hábitat (Defeo & McLachlan 2005). En ambientes estuarinos, las variaciones de salinidad son particularmente relevantes para la macrofauna, afectando su diversidad y abundancia (Lercari & Defeo 2006, Barboza *et al.*, 2012).

1.3 Amenazas para playas arenosas

Dentro de los ecosistemas costeros expuestos al triple *whammy*, las playas arenosas tienen una amenaza adicional, ya que son principalmente percibidas como áreas de esparcimiento y no como el complejo ecosistema de interfase que son (Fanini *et al.*, 2020). Como sistemas dominados por los parámetros físicos (Short, 1999; McLachlan *et al.* 1993), las playas arenosas, su diversidad y los servicios ecosistémicos que proveen están bajo un estrés incremental producido por los efectos del cambio climático (Barnard *et al.* 2015). Como motores de la economía, las playas enfrentan los efectos de la urbanización exacerbada (Short & Jackson 2013, Seto *et al.* 2012), incluyendo su fragmentación y reemplazo (McLachlan & Defeo 2018). El aumento en los números de residentes y turistas afecta negativamente estos ecosistemas, emergiendo como un problema ambiental significativo en costas urbanizadas (Reyes-Martínez *et al.* 2015, McLachlan & Defeo 2018).

Los impactos del cambio climático sobre las costas son comúnmente asociados con efectos colaterales del mismo, incluyendo el aumento del nivel del mar, el calentamiento y la acidificación del océano (Wong *et al.* 2014). El crecimiento de la economía, la población humana y la urbanización se presentan como los causantes más importantes del incremento en la exposición de las áreas costeras a los efectos del cambio climático (Defeo & Elliot 2020). Estos efectos incluyen la pérdida de tierra por erosión e inundaciones, daño por eventos extremos en ambientes construidos y pérdida de valores culturales (Revi *et al.* 2014).

Dada la alta preferencia humana por las zonas costeras, las playas representan uno de los mayores sistemas social-ecológicos del mundo, extendiéndose a través de diferentes culturas y condiciones ambientales, constituyendo un potencial modelo global para el estudio de la interacción entre la humanidad y los ecosistemas (McLachlan y Defeo 2013, Gianelli *et al.* 2015). Las áreas turísticas en general, y las playas en particular, generan dinámicas socioeconómicas particulares asociadas a la explotación de los recursos. Estas áreas cambian y evolucionan, al igual que lo hacen el perfil, las necesidades y preferencias de los visitantes (Butler 1980). En general, este proceso lleva a un deterioro gradual de las características naturales y culturales que fueron responsables por la popularidad inicial del ecosistema (Fanini *et al.* 2020), generándose la llamada paradoja del turismo y el ambiente (*tourism-environment paradox*) (Butler 1980). Los impactos humanos en los ecosistemas de playas del mundo, exacerbados por los efectos del cambio climático, hacen imprescindible una aproximación ecológica cuantificada de las relaciones entre la urbanización y estos sistemas, con el fin de asegurar la sostenibilidad de los servicios ecosistémicos que proveen.

1.4 El cambio climático y las playas de Uruguay

El Río de la Río de la Plata muestra una tendencia positiva en su caudal durante la segunda mitad del Siglo XX, debido principalmente a un aumento en las precipitaciones asociado al fenómeno de El Niño y a cambios en el uso de la tierra (Magrin *et al.* 2007). Esto tiene innumerables implicaciones sobre los ecosistemas de playas arenosas de Uruguay, destacándose la alteración de la salinidad. Lercari & Defeo (2006) encontraron una fuerte correlación entre la salinidad del agua y la diversidad de especies de macrofauna bentónica en playas de Uruguay. Una alteración en los ciclos y patrones generales de salinidad puede tener fuertes efectos sobre el macrobentos, lo que probablemente se propague a todos los componentes el ecosistema.

Los mayores niveles de inundación predichos como efecto del cambio climático para toda Sudamérica, se encuentran en el área del Río de La Plata, previéndose un aumento en la frecuencia de eventos extremos, agravado por las alteraciones en el ciclo del agua producidas en la cuenca (Magrin *et al.* 2014). El incremento del nivel del mar y el endurecimiento de la costa generan un patrón erosivo denominado compresión costera (*coastal squeeze*) que es particularmente relevante para playas arenosas (Defeo *et al.* 2009, Schoeman *et al.* 2014). El ambiente intermareal se reduce debido a la fijación del nivel dunar por desarrollo urbano costero y la construcción de infraestructuras a lo largo de la costa, mientras que el nivel inferior se eleva como resultado del aumento en el nivel del mar (Pontee 2013). A este efecto se suma el déficit de sedimento generado por las alteraciones humanas de los ciclos naturales (Short 1999), lo que agrava la reducción del área ecosistémica. Las especies de la comunidad macrobentónica de playas arenosas han mostrado fuertes preferencias de microescala por condiciones de humedad y temperatura (Celentano *et al.* 2019). De producirse una erosión significativa, las condiciones ambientales de microescala se verán alteradas, generándose impactos en la macrofauna que se propagarán a otros componentes del ecosistema, pudiendo afectar la resiliencia y los servicios ecosistémicos asociados (Jorge-Romero *et al.* 2021).

En la cuenca del Plata se han documentado variaciones interanuales y decadales en la circulación atmosférica, lo que ha llevado a detectar un aumento sostenido de la temperatura durante los últimos 40 años (Ortega *et al.* 2013). Simultáneamente se ha detectado una reducción en el número de meses secos en la temporada cálida desde mediados de los 70, mientras que aumentó la frecuencia de lluvias fuertes (Magrin *et al.* 2007). También se ha registrado un aumento en la ocurrencia de tormentas fuertes (Barreiro *et al.* 2017) con vientos de componentes sureste que acentúan el aumento del nivel del mar (Magrin *et al.* 2007, Verocai *et al.* 2015), teniendo diversos impactos ambientales y sociales en las costas de Uruguay (Magrin *et al.* 2007). Las tormentas extremas pueden erosionar y remover completamente las dunas, degradando la tierra y exponiéndola a inundaciones y cambios mayores si la recuperación del sedimento perdido no se da antes de la próxima tormenta (Short 1999). El grado de urbanización y el nivel de mar (reflejado en la altura de la ola), ambos responsables de la compresión costera, son las variables que más influyen en la recuperación de la macrofauna después de un evento de tormenta, indicando otra sinergia negativa entre cambio climático y urbanización (Machado *et al.* 2016).

Para las próximas décadas se proyecta un aumento de la erosión e inundación en la costa a nivel mundial (Bindoff *et al.* 2019). Para la costa uruguaya se han reportado patrones de variación en las características morfológicas de playas (e.g. ancho de playa, ancho de swash y pendiente) asociados principalmente con anomalías positivas en la velocidad del viento (Ortega *et al.* 2013). Varias playas urbanas de Montevideo muestran una tendencia al angostamiento, reportándose una relación entre las variaciones de área y los eventos Niño/Niña, asociada a cambios en la dirección del viento dominante (Gutiérrez *et al.* 2016). El hecho de que las comunidades dominantes en playas arenosas estén controladas físicamente (McLachlan *et al.* 1993) y que la macrofauna se componga principalmente de organismos ectotermos marino-estuarinos con alta especificidad de hábitat y baja tolerancia a los cambios de salinidad (Lercari & Defeo, 2006), los hace particularmente susceptibles a los efectos del cambio climático

(Schoeman *et al*. 2014). La variabilidad del clima, medida a través de las anomalías de temperatura superficial del mar, intensidad del viento, incremento del nivel del mar y descargas de agua dulce, ha permitido explicar tendencias de gran escala espacial y temporal en abundancia de especies, tanto en Uruguay como en Sudamérica (Ortega *et al.* 2012, Defeo *et al.* 2013).

La pérdida de terreno costero en Uruguay tendrá un potencial impacto negativo en el turismo (Magrin *et al.* 2007, Verocai *et al.* 2015). El turismo costero receptivo osciló alrededor de los 1000 millones de dólares anuales para el período 2015-2018 (Figura 2). A esto debe sumarse los valores generados por los diversos servicios ecosistémicos de apoyo (dispersión y reciclaje de nutrientes), abastecimiento (pesca de bivalvos y peces), regulación (control de los efectos erosivos de tormentas) y culturales (actividades asociadas a bienestar cognitivo). Si bien este indicador turístico es incompleto, pone de manifiesto la relevancia de desarrollar un marco teórico que incorpore los efectos del cambio climático y las presiones de explotación ejercidas sobre los ecosistemas de playas arenosas.

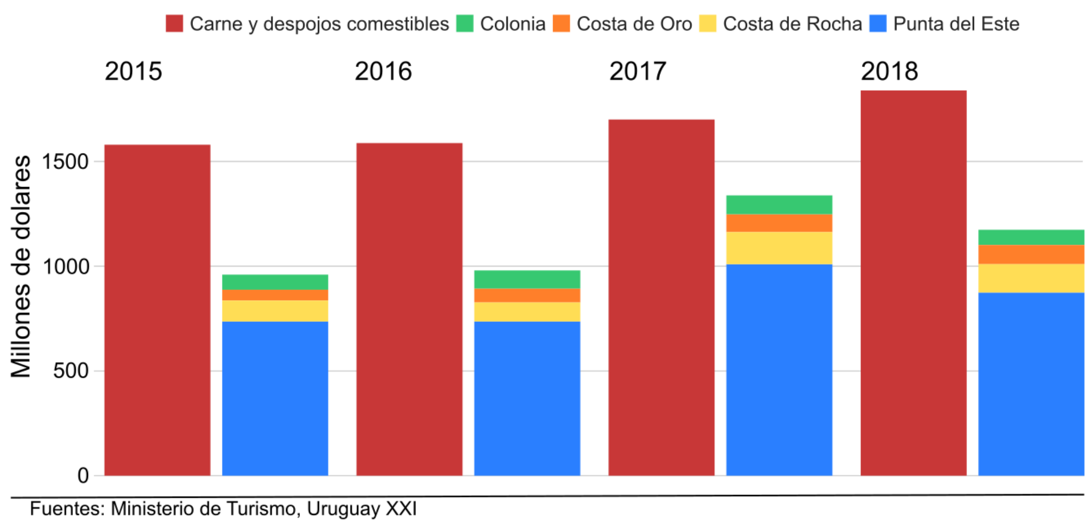

Figura 2. Millones de dólares anuales generados por turismo costero receptivo en Uruguay. La información se discrimina por localidad y se presenta el valor de exportaciones de “Carne y despojos comestibles” como referencia de la importancia del recurso natural Playa para Uruguay. Datos del Ministerio de Turismo (https://www.gub.uy/ministerio-turismo/turismoreceptivo) y Uruguay XXI (https://www.uruguayxxi.gub.uy/en/information-center/exports/).

1.5 Las playas urbanas como ecosistemas

Las playas urbanas son vistas como sitios para el turismo de arena y sol y no como el ecosistema complejo y dinámico que se desprende del conocimiento acumulado a través del estudio de playas agrestes (Defeo *et al.* 2009). Aproximadamente 40% de la población mundial vive dentro de los 100 km de la costa (UN 2014), a lo cual se suman millones de visitantes que estacionalmente visitan las playas como destino recreativo (Fanini *et al.* 2020). En consecuencia, una gran parte de la población humana está asociada, y depende económicamente, de las playas para su bienestar. Esto genera distintas dinámicas de urbanización y uso de esta infraestructura, las cuales modelan diferentes patrones de impacto que han aumentado sensiblemente en décadas recientes, resultando en alteraciones significativas de flora y fauna (Reyes-Martínez *et al.* 2015). A escala global, muchas playas han sufrido la remoción completa de la faja dunar debido a la construcción de calles y edificios, un efecto creciente de iluminación nocturna, ruidos antropogénicos, pisoteo exacerbado y contaminación de diversos tipos (Cardoso *et al.* 2016). El desarrollo costero y el turismo también afectan la estructura y funcionamiento de las redes tróficas en las playas: ambientes protegidos muestran mayor biomasa y más conexiones tróficas, reflejando sistemas más complejos, organizados y maduros en comparación con playas urbanizadas (Selleslagh *et al.* 2012).

La comunidad macro bentónica, como indicador ecológico en playas arenosas, está expuesta a disturbios directos e indirectos como resultado de la urbanización. La alteración de los ecosistemas en la cuenca de las playas, la construcción de estructuras costeras duras y la canalización de los cursos de agua, entre otros impactos urbanos, alteran los flujos de sedimento, favoreciendo los ciclos erosivos y alterando la morfodinámica (Short 1999, Jackson & Nordstrom 2011). El efecto del pisoteo humano sobre la macrofauna ha sido examinado en diversas ocasiones, encontrándose que altos niveles de pisoteo causan daños a la macrofauna (Reyes-Martínez *et al.* 2015). Además, estudios pareados han mostrado una menor resiliencia en comunidades de playas urbanas frente a eventos de tormenta cuando se compara con playas agrestes (Machado *et al.* 2016). Estos factores pueden haber llevado a una subestimación de los ecosistemas de playas urbanas, ya que debido a la baja diversidad de macrobentos se les otorga un bajo valor ecológico, sin considerar a fondo otros componentes del ecosistema.

El análisis de la urbanización es una tarea compleja. Se desprende de los ejemplos anteriores que la ecología de playas urbanas se compone mayormente de estudios que abordan el efecto de la urbanización sobre los ecosistemas mediante la comparación de ecosistemas alterados y prístinos (Machado *et al.* 2016, Selleslagh *et al.* 2012). Otras aproximaciones se basan en índices de percepción o aproximaciones locales, como contar la cantidad de usuarios en la playa (Reyes-Martínez *et al.* 2015) o utilizar información derivada de datos de viajes (Stathakis & Baltas 2018). Dada la creciente presión sobre los ecosistemas costeros, se ha llamado la atención sobre la necesidad de incorporar indicadores universales, confiables y objetivos (Cornier & Elliott 2017). Los estudios que comparan ecosistemas alterados y prístinos incorporan una incertidumbre asociada a las diferencias naturales entre los sitios. Las aproximaciones basadas en percepciones de expertos son útiles y válidas, aunque debilitan la posibilidad de hacer estudios de gran escala y realizar comparaciones, ya que los criterios varían entre áreas y expertos. Los datos de ocupación medida *in situ* presentan altos costos y complejidad logística (Short & Jackson 2003; McLachlan *et al.* 2013), mientras que los derivados de información inmobiliaria o de viajes están sujetos a la disponibilidad de información, que puede tener alta variación entre sitios.

El patrón de uso estacional de las playas urbanas presenta un desafío ecológico (Ariza *et al.* 2008; Laitano *et al.* 2019): la infraestructura es constante a lo largo del año, actuando como un disturbio de presión que puede producir impactos a largo plazo (Harris *et al.* 2018), mientras que el impacto humano directo es mayor durante la temporada de verano, funcionando como un disturbio de pulso que puede desencadenar cambios de estado ecosistémico (Jentsch & White 2019). El estudio de los sistemas social-ecológicos como las playas arenosas requiere componentes que describan cuantitativamente las dinámicas de uso humano (Ostrom 2009). Si bien el turismo recibe atención, incorporar el uso estacional de las playas al marco ecológico de manera estandarizada podría mejorar significativamente el conocimiento acerca de los impactos de la urbanización sobre los ecosistemas arenosos (Fanini *et al.* 2020).

Estudios comparativos que cuantifiquen la urbanización y su uso, y los incorporen como variables explicativas, tienen el potencial de mejorar el conocimiento sobre el funcionamiento y los servicios asociados a ecosistemas de playas, tanto prístinas como urbanas. Evaluar los impactos de la urbanización y la creciente demanda de actividades recreativas en playas, sumado a los efectos del cambio climático, puede brindar fundamentos para la implementación de planes de manejo tendientes a mejorar la calidad de vida en urbes costeras y aumentar la conciencia sobre la importancia de preservar los ecosistemas de playas.

2. OBJETIVOS

La actividad humana a nivel planetario es probablemente el factor más influyente sobre los ecosistemas, y será aún más relevante en el futuro. La urbanización y el uso de los ecosistemas deben ser incorporados en las prácticas científicas de manera estandarizada y comparable. Esto resulta imprescindible para el desarrollo de marcos teóricos que aborden las relaciones entre ecología y urbanización a escala local y global. El desarrollo urbanístico, con sus diversas expresiones de alteración, fragmentación y explotación de los ecosistemas, se concentra en áreas costeras asociadas a playas arenosas. Sumado a esto, los efectos del cambio climático se evidencian en las playas a través de un aumento en la temperatura atmosférica y del mar, así como en el incremento del nivel del mar y la frecuencia de eventos climáticos extremos. En este contexto, en esta tesis se abordó el desarrollo de un marco analítico para playas urbanas apuntado a fortalecer la comprensión de las relaciones entre los ecosistemas de playas y la urbanización.

2.1 Objetivo general

Evaluar los efectos de la urbanización sobre los ecosistemas de playas arenosas, incorporando estimadores satelitales ecosistémicos y antrópicos al marco de la ecología de playas. Se espera que la inclusión de indicadores cuantitativos disponibles a escala global contribuya a mejorar las aproximaciones ecológicas y de manejo en playas, permitiendo un análisis más detallado de los impactos de la urbanización y el uso humano, así como el cambio y la variabilidad climática sobre estos ecosistemas.

2.2 Objetivos específicos

1) Analizar los efectos de distintos forzantes climáticos sobre el área de los ecosistemas de playas del departamento de Montevideo, y discutir sus potenciales implicaciones en el contexto del cambio climático.

2) Analizar el efecto de diferentes indicadores urbanos sobre la comunidad macrobentónica de playas arenosas.

3) Contrastar y complementar recomendaciones de expertos para el manejo de playas arenosas con aproximaciones basadas en indicadores satelitales, haciendo énfasis en la urbanización y su uso estacional.

3. ESTRATEGIA DE INVESTIGACIÓN

Las playas arenosas son sistemas dinámicos muy vulnerables al impacto humano, que además reciben una fuerte exposición al mismo. Las playas urbanas padecen aún más este problema, ya que son percibidas como extensiones de la ciudad, relegando la condición de ecosistema justamente donde existen mayores presiones antrópicas y mayor potencial de aprovechamiento de servicios ecosistémicos locales. Para valorizar los ecosistemas de playas urbanas debe formalizarse su estudio, incorporando y cuantificando la variable dominante en sistemas social-ecológicos del Antropoceno: la actividad humana. Hasta el momento, se han utilizado principalmente aproximaciones basadas en índices de percepción, las cuales limitan la capacidad de trabajar a macroescala. Es necesaria una aproximación cuantitativa para abordar los desafíos inherentes a estos sistemas complejos.

En el contexto de creciente presión antrópica costera, las playas urbanas pueden ser consideradas como un experimento de la situación en la que eventualmente estará la mayoría de las playas del mundo. En este sentido, avanzar en la cuantificación de las variables que representan la urbanización para lograr un abordaje más formal tiene implicaciones sobre la ecología urbana, la ecología de playas y el manejo costero. Esta tesis aborda esta temática desde una postura pragmática, trabajando con variables que puedan representar características ecosistémicas, así como grados de urbanización y uso humano, para explorar sus relaciones con variables ecológicas y decisiones de manejo. El trabajo se estructura sobre 3 objetivos específicos que abordan, de manera complementaria, las relaciones entre forzantes antropogénicos y los ecosistemas de playas arenosas. A cada objetivo le corresponde un artículo científico y un capítulo de esta tesis (Tabla 1).

En el primer artículo (Capítulo 4) se analiza el comportamiento del área de las playas de Montevideo en el largo plazo, y su relación con las variables climáticas. Este análisis tuvo dos etapas: una inicial de estimación del área de playa mediante la aplicación de algoritmos de clasificación en fotografías satelitales; y otra posterior en la cual se utilizó la información del área de playa a efectos de relacionarla con distintos forzantes climáticos. Esto permitió generar conocimiento sobre la dimensión física de las playas y su comportamiento, y además contrastar los efectos del cambio climático sobre estos ecosistemas. La metodología desarrollada para este análisis está basada en información abierta, la cual se detalla en el artículo incluido en el Anexo I.

En el segundo trabajo (Capitulo 5) se incorpora el factor urbanización a un estudio clásico de ecología de playas en Uruguay, a través de colecciones históricas de fotografías satelitales, densidad humana y luces nocturnas. Se evaluó la relevancia de la urbanización como variable explicativa de la riqueza de especies macrobentónicas. Para esto se utilizó una aproximación de modelos lineales múltiples, los cuales facilitaron la comparación del factor urbanización con los resultados preexistentes, y permitieron dimensionar su importancia en el contexto de la ecología de playas.

En el tercer trabajo (Capítulo 6) se aborda la importancia de los patrones de uso humano y sus consecuencias sobre los ecosistemas de playas, en el entendido de que en estos sistemas social-ecológicos se debe incorporar la actividad humana de manera cuantificada y comparable. Se trabajó a escala global analizando las correlaciones entre variables satelitales con un índice de manejo basado en la opinión de expertos.

Tabla 1. Resumen de los objetivos específicos, las publicaciones asociadas y la sección correspondiente de este documento.

Objetivo específico resumido	**Referencia de la publicación**	**Capítulo**
Analizar efectos de forzantes climáticos sobre el área de los ecosistemas de playas del departamento de Montevideo	Orlando, L., Ortega, L., Defeo, O., 2019. Multi-decadal variability in sandy beach area and the role of climate forcing. Estuarine, Coastal and Shelf Science 218, 197–203.	4. Variabilidad multidecadal del área de playa, el rol de los forzantes climáticos
Determinar el efecto de la urbanización sobre la comunidad macrofaunística de playas arenosas	Orlando, L., Ortega, L., Defeo, O., 2020. Urbanization effects on sandy beach macrofauna along an estuarine gradient. Ecological Indicators 111, 106036.	5. Efecto de la urbanización sobre la macrofauna en playas a lo largo de un gradiente estuarino
Contrastar y complementar recomendaciones de expertos para el manejo de playas arenosas, con aproximaciones basadas en indicadores satelitales cuantificados	Orlando, L., Ortega, L., Defeo, O. Perspectives for sandy beach management in the Anthropocene: satellite information, tourism seasonality, and expert recommendations. *Enviado a: Estuarine Coastal and Shelf Science.*	6. Perspectivas para el manejo de playas en el Antropoceno: información satelital, estacionalidad turística y opinión de expertos

4. VARIABILIDAD MULTIDECADAL DEL ÁREA DE PLAYA: EL ROL DE LOS FORZANTES CLIMÁTICOS

4.1 Resumen

Las playas arenosas comprenden el 31% de las costas libres de hielo del mundo y brindan una variedad de servicios ecosistémicos esenciales para el bienestar humano, en particular la protección contra la erosión y los fenómenos climáticos extremos. Estos ambientes altamente dinámicos están controlados principalmente por la energía de las olas, las mareas y el tamaño de grano, generando patrones morfodinámicos complejos que se traducen en una fuerte variación del área de playa. Las estimaciones de dicha área y su variabilidad son insumos primordiales para la gestión costera, un desafío que requiere datos que representen un amplio rango espacio temporal. En este artículo se desarrolló una metodología semi automatizada basada en información de la colección Landsat, para reconstruir el área de 21 playas arenosas de la costa de Montevideo (Uruguay) entre 1984 y 2016. Esta información de largo plazo se utilizó para discernir los ciclos de erosión-acreción por medio del análisis de ondículas y para explorar el papel de en los forzantes climáticos a través de modelos lineales mixtos. Las tendencias a largo plazo mostraron un ciclo de 27 años con fases de erosión y acreción cuasi-decadales relacionadas con diferentes configuraciones climáticas. El área de playa fue afectada negativamente por el aumento del nivel del mar y las condiciones climáticas del año anterior, estando correlacionada positivamente con anomalías positivas de la temperatura superficial del mar y vientos de tierra (que favorecieron la acreción) y negativamente con vientos desde el mar y eventos intensos de El Niño (que favorecieron la erosión). Estos resultados, junto con el aumento previsto en la ocurrencia e intensidad de tormentas y eventos extremos de El Niño, constituyen un escenario preocupante, ya que la erosión en esta zona costera poblada podría tener efectos sociales, económicos y ecológicos negativos. La metodología desarrollada en este trabajo fue útil para detectar cambios a largo plazo en el área de la playa y está basada completamente en información de acceso abierto. Por lo tanto, es potencialmente aplicable en cualquier lugar del planeta. Este enfoque también es útil para contrarrestar la escasez de información a largo plazo que ha impedido una evaluación sólida de los efectos del cambio climático en las zonas costeras.

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Multi-decadal variability in sandy beach area and the role of climate forcing

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ABSTRACT

Sandy beaches comprise 31% of the ice free coasts of the world, providing a variety of ecosystem services essential to support human well-being, particularly protection against erosion and extreme climatic events. These highly dynamic environments are controlled principally by wave energy, tides and grain size, generating complex morphodynamic patterns that translate into strong area variation. Estimates of beach area and its variability are prime inputs for coastal management, a challenge that requires data representing a wide spatiotemporal scale. In this paper, a Landsat-based semi-automated methodology was developed to reconstruct the area of 21 sandy beaches of the Montevideo (Uruguay) coast from 1984 to 2016. This long-term information was also used to discern erosion-accretion cycles by means of wavelet analysis, and to explore the role of climatic forcing on these cycles through linear mixed models. A random forest classification algorithm was applied to Landsat images in order to estimate beach area. Long-term trends described a 27-year cycle with well-delimited quasi-decadal erosion and accretion phases related to climatic configurations. The beach area was negatively affected by an increase in sea level and climatic conditions of the previous year, being positively correlated with sea surface temperature anomalies and offshore winds (which favored accretion) and negatively correlated with onshore winds and intense El Niño Southern Oscillation (ENSO) events (favoring erosion). Our findings, together with the predicted increase in the occurrence and intensity of storms and extreme ENSO events, constitute a worrying scenario, as erosion in this populated coastal zone could have negative social, economic and ecological repercussions. The methodology developed here was useful to detect long-term changes in beach area and is entirely based on open-access information. Therefore, it is potentially applicable at any location on the planet. This approach is also useful to counteract the scarcity of long-term information that has precluded robust assessments of climate change effects on coastal zones.

1. Introduction

Sandy beaches comprise 31% of the ice free coasts of the world (Luijendijk et al., 2018), and provide a wide variety of ecosystem services essential to support human well-being. These services include, among others: sediment storage and associated buffering against extreme events (Short, 1999); water filtration and purification (Constanza et al., 2014); and maintenance of biodiversity and economic development for humans (Defeo et al., 2009). The physical structure of sandy beaches is determined by the interaction between sand, waves and tides (Short, 1999). Sediment transport, in the surf zone by wave action and in the dunes by wind action, promotes an extremely dynamic environment where sand and water are always in motion. From a two-dimensional perspective, this high dynamics translates into strong short-term beach area variations (Short, 1999).

Ecosystem area is a major ecological variable, positively correlated with species diversity, population persistence and food chain length (Schoener, 1989; Takimoto and Post, 2013), and could also impact ecosystem attributes such as resilience (Dunne et al., 2002) and services (Balvanera et al., 2014). In sandy beach ecosystems, studies drawn from a pool of widely differing beaches all around the world have shown a consistent increase in species richness, abundance and biomass of macrofaunal communities as beach width increases (reviewed in McLachlan and Defeo, 2018). It has been also shown that beaches with larger intertidal areas support larger populations, thus reducing the probability of species extirpations (Defeo and McLachlan, 2013; Barboza and Defeo, 2015; Defeo et al., 2017). Therefore, habitat availability, as indicated by beach width, may be important in explaining local to regional variations in sandy beach diversity patterns. Stressors that affect beach area also impact over ecosystem attributes, especially those associated with resilience to natural hazards (Defeo et al., 2009), seriously undermining human well-being in areas where

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population density is nearly three times higher than the global average (Small and Nicholls, 2003).

Coasts are undergoing increasing use by human populations worldwide, characterized by an intensification of urban development (Defeo et al., 2009). Sandy beaches are particularly vulnerable to physical modifications derived from sand mining, armoring structures and urban-associated alterations, which reduce the ecosystem area and alter sediment budget and cycles, thus accelerating erosion rates (Short, 1999; Torres et al., 2017). Global climate change, particularly sea-level rise and increased storm frequency, has added a new dimension to worldwide modifications of shorelines (McLachlan and Defeo, 2018). In this context of climatic and human stressors converging on coasts, 24% of the world's sandy beaches are persistently eroding (Luijendijk et al., 2018).

Estimations of beach area and its temporal variability are prime inputs for coastal management (Short and Jackson, 2013); a challenge that requires data representing a wide range of spatial and temporal scales (Barnard et al., 2012). Coastal monitoring on-site programs provide insights on coastal dynamics (Harley et al., 2010), but are expensive and often temporally and spatially sparse (Splinter et al., 2013). This is especially problematic, as robust long-term data sets are required before meaningful trends emerge (Harley et al., 2010). Therefore, several information sources for shoreline analysis and beach area estimation have been explored to complement on-site surveys (Short and Jackson, 2013). Among them, aerial photographs provide excellent spatial coverage of beach systems, but are heavily limited in their time and geographic coverage (Short and Jackson, 2013). Satellite imagery has been highlighted as the most realistic way forward in providing useful data for studies of nearshore morphodynamics (Harris et al., 2011; Short and Jackson, 2013) and has been widely used for shoreline and coastal monitoring at global (Luijendijk et al., 2018) and local (Harris et al., 2011; Ozturk and Sesli, 2015) scales. Landsat satellite series have spatial, spectral, and radiometric resolution and, along with their temporal continuity, have proven well-suited for land characterization activities (Ozturk and Sesli, 2015, Luijendijk et al., 2018).

Montevideo city hosts approximately 45% of the total population of Uruguay, with the most conspicuous residential areas, industries and services (e.g. oil refinery, port, recreational areas) concentrated within its coastal zone (Saizar, 1997). Besides its ecological and economic importance, the coast is the most iconic public space and an integral part of Montevidean identity and daily life. Due to its recreational use, natural characteristics, architectural values, ethnical links to immigrant culture and even spiritual connotations, it has been proposed as a World Heritage site (UNESCO, 2010). However, the Uruguayan coast and its sandy beaches have been increasingly affected by climate change stressors, including sea-level rise (Fig. 1b) and warmer sea waters (Fig. 1c), which have been accompanied by flooding/drought events and an increase in the frequency of storms and in the frequency and intensity of onshore winds (Ortega et al., 2013; Barreiro, 2017). Altogether, these stressors could affect sandy beach ecosystems, which can be perceived as ecosystems at risk. There is thus a need to increase knowledge on long-term sandy beach dynamics, including variations in erosion and accretion phases and their relationship with climate forcing (Barnard et al., 2015).

Semi-automated classification of information has been extensively applied on remote sensing studies as a way to map different ecosystems land cover at local and global levels (Millard and Richardson, 2015; Hansen et al., 2013). This methodology requires reference training areas for the classifying algorithm to map the categories on the region of interest (Millard and Richardson, 2015). In this paper, a Landsat-based semi-automated methodology was developed to reconstruct the area of 21 sandy beaches of the Montevideo coast from 1984 to 2016. This long-term information is also used to discern erosion-accretion cycles by means of wavelet analysis, and to explore the role of climatic forcing on these cycles through linear mixed models.

2. Methods

2.1. Study site

The Montevideo coast (Fig. 1) is characterized by sandy beaches interrupted by rocky heads, with a semidiurnal tidal regime with microtidal amplitude (ca. 0.5 m). However, strong onshore winds produce unpredictable short-term increases in sea level (storm surges up to 4 m) (Lercari and Defeo, 2006). The Rio de la Plata has a major freshwater input from Parana and Uruguay rivers, forming a shallow (up to 15 m) coastal-plain estuary (Lercari and Defeo, 2006). A strong turbidity front is located around Montevideo city, which constitutes the surface indication of the transition between fresh and saline waters (Sepúlveda et al., 2004). In this estuary, variability of water characteristics (salinity, temperature and turbidity) is mainly forced by winds (Simionato et al., 2010).

2.2. Beach area estimation

Beach area was estimated from 1984 to 2016 for 21 beaches of the Montevideo coast (Fig. 1S, Supplementary Material). Almost all beaches included in this study are southern-oriented (10 towards the S, 5 to SE, 5 to SW and 1 to W) (following Engstrom, 1973). Google Earth Engine (GEE) open platform for satellite image processing (Gorelick et al., 2017) and associated satellite collections were used. Beaches were measured using all the available standard Level 1 Terrain-corrected orthorectified images from Landsat 5 (L5) (1984–2011) and Landsat 7 (L7) (2004–2016). All satellite information was restricted to a 20-km radius of coordinates 34°52′12″S, 56°13′48″W (Fig. 1a). L5 had an average of 15 images per year with a minimum of 6 (1985) and a maximum of 31 (2004), while L7 had an average of 32 images per year with a minimum of 28 (2010) and a maximum of 38 (2011). Landsat images were converted to top of the atmosphere reflectance to remove the effect of clouds and shadows (Luijendijk et al., 2018). A median-based annual composite was constructed from the non-cloudy pixels and used to estimate beach area. In order to improve the accuracy of beach area estimates, the water layer was removed by applying a normalized difference water index (NDWI) mask (Gao, 1996), combined with photo interpretation. Polygons enclosing each of the 21 beaches were constructed based on Quickbird satellite images from Google Earth (2.5 m spatial resolution), and used to obtain individual measures for each beach. A random forest supervised classification algorithm was applied to distinguish sand from other kinds of cover (e.g., rocky, hard structures, vegetation), the 7 (8) spectral bands of Landsat 5 (7) were considered as classification variables (Cutler et al., 2007). For the sand category, training polygons were distributed along the coast, and sand areas were interpreted from imagery and relocated for each year composite to deal with variations on shoreline position and image characteristics. The category "other" was formed by polygons that comprised all non-sand cover, including urban, rocky, rural and vegetation cover. For each category, 100 training points were randomly extracted from the training areas, considering that: 1) the number of training points should be at least ten times the number of variables used in the classification (Jensen, 2005); and 2) random selection of points within training areas reduces spatial autocorrelation and classification errors (Millard and Richardson, 2015). As sand only represents 0.01% of the total surface in the study region, collecting proportionally more training data can enable a better representation of the statistical characteristics and variability of these proportionately-smaller classes, even though it may lead to over-representation (Millard and Richardson, 2015). In order to override this factor and provide a better characterization of the sand category optical properties, the threshold of the majority vote to determine the predicted class of every pixel was increased to 65% of the predictions for sand observations. Following this procedure, 1000 decision trees were constructed for each year. All pixels within the beach polygons and classified as "other" with a

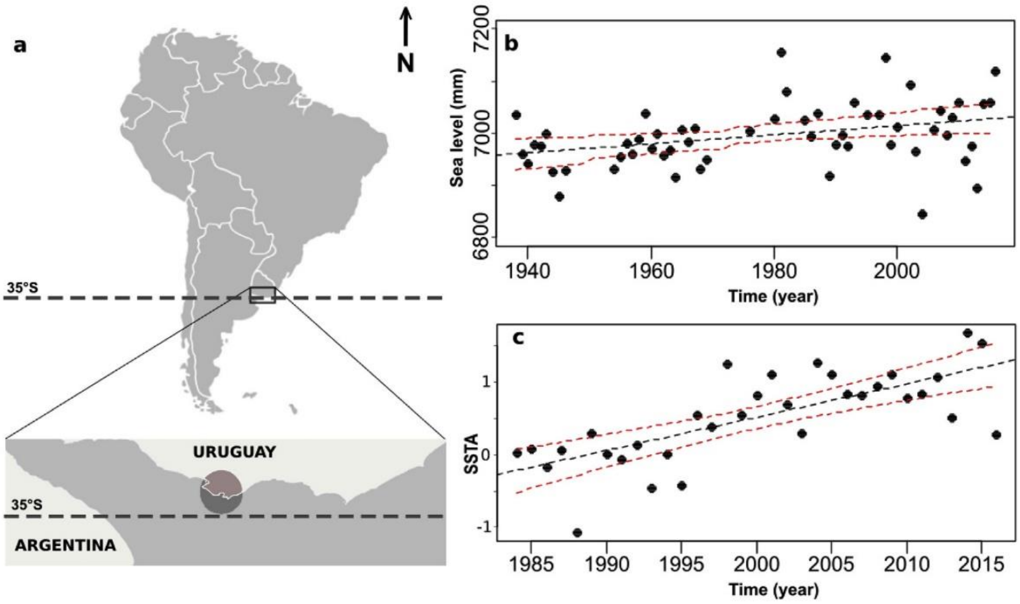

Fig. 1. a) Study area, with the circle in the lower panel highlighting a 20-km radius area that includes the coast of Montevideo where this study was focused. Long-term trends with linear models and 95% confidence intervals in b) annual mean sea level in Montevideo ($y = 0.875x + 5263.1$, $R^2 = 0.12$); and c) yearly sea surface temperature anomalies (SSTA) for the region (40W-60W/30S-40S) ($y = 0.046x - 91.261$, $R^2 = 0.51$). Both trends are highly significant ($p < 0.01$).

Table 1
Summary of climatic variables used in this study, detailing units, a general description and information source.

Variable	Units	Description
Meridional Wind	m/s	Annual mean of monthly meridional winds for the region (40W-60W/30S-40S)
Zonal Wind	m/s	Annual mean of monthly zonal winds for the region (40W-60W/30S-40S)
Wind direction	Degrees	Direction of resultant wind from Zonal and Meridional
Wind velocity	m/s	Speed of resultant wind from Zonal and Meridional
Categorical wind direction	Categorical	Categorization of wind direction in 8 groups (N, NE, E, …)
Sea level Montevideo	mm	Level of the sea measured in the Montevideo Port
Global Sea level	mm	Global level of the sea
Precipitation anomaly in Montevideo	mm	Anomaly of precipitations in Montevideo
Precipitation anomaly in the de la Plata river basin	mm	Anomaly of precipitations at the Rio de la Plata basin
Atmospheric temperature	°C	Anomaly of temperature at 2 m of altitude (40°W-60°W/30°S-40°S)
Sea surface temperature anomaly (SSTA)	°C	Anomaly of sea surface temperature (40°W-60°W/30°S-40°S)
Uruguay river flow anomaly	m^3/s	Anomaly of water flow from Salto Grande Hydrological power plant
Rio de la Plata flow anomaly	m^3/s	Anomaly of water flow from Instituto Nacional del Agua y el Ambiente
ENSO 3.4	Continuous	SSTA based index for determining Niño/Niña events and intensity
ENSO	Categorical	ONI based classification of years as Niño, Niña or neutral, and intensity
Pacific Decadal Oscillation (PDO)	Continuous	Yearly average of PDO monthly values
Atlantic Multidecadal Oscillation (AMO)	Continuous	Yearly average of AMO monthly values

normalized difference vegetation index > 0, were considered as beach vegetation and added to the total beach area. To allow comparisons between beaches with different sizes, the normalized beach area was calculated as the standard score for each beach (standard score (i) = (i-mean)/standard deviation), and will be referred as beach area hereafter.

2.3. Wavelet and coherence analysis

Wavelet transform is a well-suited tool for the study of non-stationary processes occurring over finite spatial and temporal domains (Lau and Weng, 1995) and has been applied to analyze temporal patterns of beach erosion (Short and Trembanis, 2004). By decomposing a time series into a time-frequency space, it is possible to determine both the dominant modes of variability and their variation through time (Torrence and Compo, 1998). In order to evaluate significant cyclic behavior in beach area and to determine its frequency, a wavelet analysis was applied using the "dog" mother wavelet (Short and Trembanis, 2004). Coherence analysis was performed to complement wavelets by measuring the cross-correlation of two time series as a function of frequency (Torrence and Compo, 1998), which can be interpreted as a decomposition of the correlation coefficient at a different scales, i.e., the local correlation between time series in a time-frequency space (Casagrande et al., 2015). This analysis reveals the interaction between different processes at different scales, and points out which variable leads over the other. Coherence analysis was applied to beach area against El Niño Southern Oscillation (ENSO) 3.4 index, Atlantic Multidecadal oscillation (AMO) and the Pacific Decadal Oscillation (PDO) index, using the "morlet" mother wavelet (Torrence and Compo, 1998). All wavelet and coherence analyses were made using the *biwavelet* package (R Development Core Team, 2012).

2.4. Climatic information and modeling procedure

In order to assess the relationships between beach area estimations and climate, annual averages of climatic variables linked to beach erosion/accretion (Short, 1999), and those relevant for the region (Ortega et al., 2013) were gathered (Table 1). Yearly-average sea surface temperature anomalies (SSTA) were estimated from a 1° × 1° latitude, longitude resolution grid and five wind variables were assessed: meridional and zonal wind velocity (IRI/LDEO, Reynolds et al., 2002; Kalnay et al., 1996), and the resultant wind direction and speed and categorized wind direction (CWD) in 8 orientations (N, NE, E, SE, S, SW, W, NW). Two measures of sea level were included, one local and one global (GLOSS, Bradshaw et al., 2015). Temperature was considered at 2 m high in the atmosphere and in the sea surface (NCEP-NCAR, Kistler et al., 2001). As precipitation and river flow have been linked with sea level and erosive patterns in the Uruguayan coast (Saizar, 1997), precipitations at site and in the Rio de la Plata basin (PERSIANN CDR, Sorooshian et al., 2014) were included. River flow estimates for the Rio Uruguay and Rio de la Plata (Borus et al., 2017) were also included. To represent general climatic patterns, global climatic indices were considered: the PDO continuous index (NCEI/NOAA, Mantua and Hare, 2002), the AMO index (NOAA/NCEP, Enfield et al., 2001) and the ENSO 3.4 continuous index (NOAA/NCEP CPC, Trenberth, 1997). Changes in the PDO and AMO tend to coincide with changes in the relative frequency of ENSO events affecting precipitations, SST and atmospheric patterns in the study region (Andreoli and Kayano, 2005; Verdon and Franks, 2006). ENSO 3.4 has shown the highest correlation with precipitations, SST and atmospheric pressure anomalies on the South Atlantic (Colberg et al., 2004). Also, an ENSO categorical index based on the Oceanic Niño Index (ONI) was included, with events defined as 3 consecutive overlapping 3-month periods at or above the +0.5° anomaly for warm (El Niño) events and at or below the −0.5 anomaly for cold (La Niña) events. This threshold was further broken down into Weak (with a 0.5 to 0.9 SSTA), Moderate (1.0–1.4), Strong (1.5–1.9) and Very Strong (≥2.0) events (GGWS, Null, 2018).

Linear mixed models were used for modeling the relationship between beach area and climatic variables. To assess long-term trends in erosion-accretion patterns in the Montevideo coast, the normalized areas of each of the 21 beaches were taken as replicates within each year. The incorporation of random effects allowed us to combine variations in annual climatic configuration with direct effect variables acting on beach area (Bolker et al., 2009). The categorical ENSO variable was included as random intercept in order to summarize possible effects on beach area related to different ENSO conditions. Pearson's correlation coefficient was applied to discard redundant variables, and then linear mixed models were started with different combinations of variables and refined by dropping non-significant variables, until models including all significant variables were accomplished. Akaike's Information Criterion (AIC) was applied to evaluate each model's fit and parsimony, as well as for selecting the best model. All climatic variables were modelled with and without a 1-year time lag, on the understanding that current beach area is the result of a temporal integration of forcing factors. For the random intercept, lag-levels were determined through coherence analysis. Linear modeling assumptions were tested (Bolker et al., 2009) and modeling procedures were performed applying the *lmer* package of R.

3. Results

A total of 693 measures were obtained from 21 beaches over a 33-year period (1984–2016) (Fig. 2a). Beach area showed a cyclic long-term pattern with phases of accretion (positive beach area variation) and erosion (negative beach area variation). Global wavelet spectrum showed a significant dominant periodicity of 27 years (Fig. 2b), which is consistent with the observed multidecadal cycle in the beach area (Fig. 2a). Wavelet variance of beach area decreased through time, following the cycle of accretion/erosion transitions (Fig. 2c). Coherence analysis showed a statistically significant relationship between the ENSO 3.4 index and beach area at periods of 1–3.5 years between 1997 and 2003, where beach area lags ENSO (Fig. 2S, Supplementary Material). No significant relationships were found between beach area and the multidecadal climatic indexes, PDO and AMO.

The best fit model for beach area (Table 2, Fig. 3) included a negative effect of sea level (Fig. 3b), a positive effect of SSTA recorded in the previous year (Fig. 3c) and an effect of 1-year lagged categorical wind direction. The positive effect over beach area was associated with northerly winds, while winds with a south component had a negative effect (Fig. 3d). One-year lagged Categorical ENSO showed the best results, with an overall negative effect on beach area (Fig. 3e). Only weak La Niña years showed a positive influence on beach area, with erosive effects increasing with the intensity of the events.

4. Discussion

Long-term trends in sandy beach area at Montevideo described a 27-year cycle, with well-delimited quasi-decadal erosion and accretion phases related with climatic configurations. The modeling process selected simultaneous and lagged annual variables to explain long-term patterns in beach area. The statistically significant relationships between beach area and current (e.g., sea level) and past (1-year lagged mean wind direction, SSTA and ENSO index) forcing conditions reinforce the notion that drivers acting at different time scales are co-responsible for the present and future behavior of sediment budget in sandy beaches (Short, 1999). It also highlights the role of the past climate forcing in erosion/accretion cycles.

The methodology developed here was useful to detect long-term changes in beach area and is entirely based on open-access information. Therefore, it is potentially applicable at any location on the planet, particularly in those regions where the regional wave climate genesis, tidal regime, and sediment transport regimes are comparable. This approach could be used to counteract the scarcity of long-term information that has precluded robust assessments of climate change effects on coastal zones (Harley et al., 2010; Harris et al., 2011; Short and Jackson, 2013; Splinter et al., 2013). However, its implementation should be conducted with caution, particularly in cases of differences in effective Landsat coverage (which could vary according to location). Close collaboration with local sandy beach researchers is strongly emphasized. In general, satellite-based studies of coastal behavior are mainly directed to determine the position of the water/sand interface and are mainly focused on erosion rates (Ozturk and Sesli, 2015; Luijendijk et al., 2018). In this context, the methodological approach developed here was useful to reconstruct information on coastal behavior and, at the same time, allowed for the detection of sand and vegetation areas within the beach, as well as their distribution in space and time, a feature with relevant potential for ecological studies and conservation and management programs. Indeed, the approach could also be useful to determine the area of planning units (beaches), and to integrate information from different beaches on a large scale to assess variations in sand budget and ecosystem quality, which in turn could be critical to determine reserve networks directed to sustain biodiversity (Margules and Pressey, 2000). In addition, the methodology is suitable to help ensure the sustainability of ecosystem services associated with sandy shores (Harris et al., 2011). Different management actions should be incorporated in the modeling process, taking into account that changes to beaches and dunes due to both unplanned and planned human actions (e.g., nourishment, engineering structures such as groins, revetments and breakwaters) are increasing exponentially, reducing sediment supply and therefore altering beach area.

The positive relationship between beach area and lagged SSTA could be explained by the integrated role of wind patterns, oceanic circulation, precipitations and solar insolation (Simionato et al., 2010), that took place the year before. Therefore, SSTA could be considered as

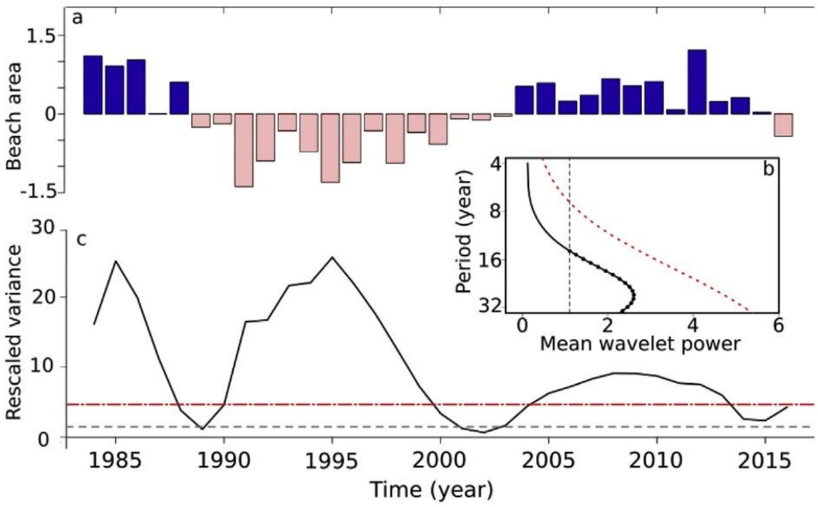

Fig. 2. Wavelet analysis. a) Annual median normalized beach area over the study period (1984–2016). b) Global wavelet spectrum: the vertical dashed line is the white noise threshold, the curved dashed line is the red noise threshold, and the mean wavelet power (variance) is the continuous black line with significant periods represented by dots. c) Scaled average power (mean variance) in time (solid line). The red dot-dash line represents 95% red noise threshold and the grey dashed line shows 95% white noise threshold. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 2
Linear mixed models adjusted for predicting normalized beach area. Fixed effect variables and random effects variables on the model intercept are listed for each model. **d.f.**: degrees of freedom, **AIC**: Akaike information criterion, $\mathbf{R^2m}$ marginal R^2, $\mathbf{R^2c}$ conditional R^2. Sea level was measured at Montevideo Port; wind was categorized in eight directions; SSTA: sea surface temperature anomaly; ENSO: El Niño Southern Oscillation, discriminated by category according to phase and intensity; RdlP: Rio de la Plata; PDO: Pacific Decadal Oscillation index; "*lag*" indicates 1-year lagged variable.

#	Fixed effects	Random effect	d.f.	AIC	R^2m	R^2c
1	sea level + categorized wind direction *lag* + SSTA *lag*	ENSO *lag*	12	1858	0.18	0.40
2	sea level + meridional wind *lag* + SSTA *lag*	ENSO *lag*	6	1868	0.10	0.20
3	categorized wind direction *lag* + SSTA *lag* + RdlP flow *lag*	ENSO *lag*	12	1886	0.17	0.44
4	meridional wind *lag* + SSTA *lag* + RdlP flow	ENSO *lag*	6	1890	0.10	0.17
5	Uruguay flow *lag* + meridional wind *lag* + SSTA *lag* + PDO lag	ENSO *lag*	7	1903	0.11	0.23

an aggregate variable that carries different simultaneous effects of the previous year general conditions in current beach state. Higher SSTA values are associated with warm calm conditions in the study area (Simionato et al., 2010) that favor accretion of beaches, whereas lower SSTA has been associated with strong onshore winds (Alves and Pezzuto, 2009) that produce high-energy waves inducing erosion. This is in agreement with the negative effects of southerly wind on beach area estimated by the linear mixed model (Fig. 3d) and the Montevideo coast orientation, which is mainly south-facing (Fig. 1). Offshore (north) winds pushes the water away from the coast and decrease wave energy, favoring accretion, whereas onshore (south) winds increases the aerial loss of sediment to the land and wave energy, also pushing the water over the beach, resulting in augmented erosion rates. Wind direction is therefore an important explanatory variable in the evolution and orientation of coastal features (Bauer et al., 2012), and its effects on beach area can be ascribed to aerial erosion and a modulation of wave energy.

The ENSO signal influences SSTA and air temperature in the region, generating complex climatic configurations that modify precipitation and wind patterns (Barreiro, 2010) and therefore altering beach area. The increase of northerly winds during El Niño (Barreiro, 2017) has been invoked to explain an increase of beach area in the region (Gutiérrez et al., 2016). However, increasing precipitation levels and storm occurrence with El Niño intensity resulted in increased erosion (Fig. 3e). Higher incidence of strong winds from the south quadrant during La Niña (Barreiro, 2017) also increased erosion. However, there is a decrease in precipitations after weak La Niña events, and the erosion by wind is likely to be compensated by calm climatic conditions that favor accretion (Gutiérrez et al., 2016) (Fig. 3e). In that vein, the period between 1998 and 2002 was characterized by a dominance of weak and moderate La Niña (Null, 2018), when beach area tended to increase, followed by a shift to a positive accretion phase.

The erosion/accretion patterns in Montevideo beaches responded to physical forcing and changing climatic conditions, supporting the shared concern about the potentially critical effects of climate change in coastal areas (Barnard et al., 2015). The sea surface temperature of the South Atlantic Ocean is expected to increase linearly at a decadal scale (Fei-Fei et al., 2012). A doubling in the frequency of extreme El Niño events, followed by extreme La Niña, has been predicted (Cai et al., 2015), with potentially negative effects on sandy beaches in the region. According to the modeling results presented here, the enhanced intensity and frequency of ENSO events, together with sea level rise and increased storminess, constitute a highly erosive scenario for the Montevideo coast. This has utmost importance, taking into account that this coast is influenced by waters of the Southwestern Atlantic Ocean, a major global hotspot where warming occurs at several times the average global rate (Hobday and Pecl, 2014). Indeed, the warm front in this region, as indicated by the 20 °C isotherm, showed a consistent long-term poleward shift at a rate of $9\,\mathrm{km\,yr^{-1}}$ and coastal air temperatures in the region are also warming (Ortega et al., 2016).

In summary, the modeling approach selected global (ENSO) and regional (SSTA) climatic indices, in combination with wind direction (a proxy for wave energy and direction) and sea level as relevant correlates of beach area. The set of selected variables has a similar structure to the findings of Barnard et al. (2015) for the Pacific Ocean basin, where climatic indices, wave energy flux, wave direction, and sea level anomalies were strongly related to coastal response. The positive long-term relationship between temperature (i.e., SSTA) and sea level, together with the occurrence of more frequent and severe storms resulting from warmer air and sea temperatures (Fig. 1c) is threatening Montevideo beaches and could increase erosion rates, a trend already observed in sandy beaches of the Atlantic coast of Uruguay (Ortega et al.,

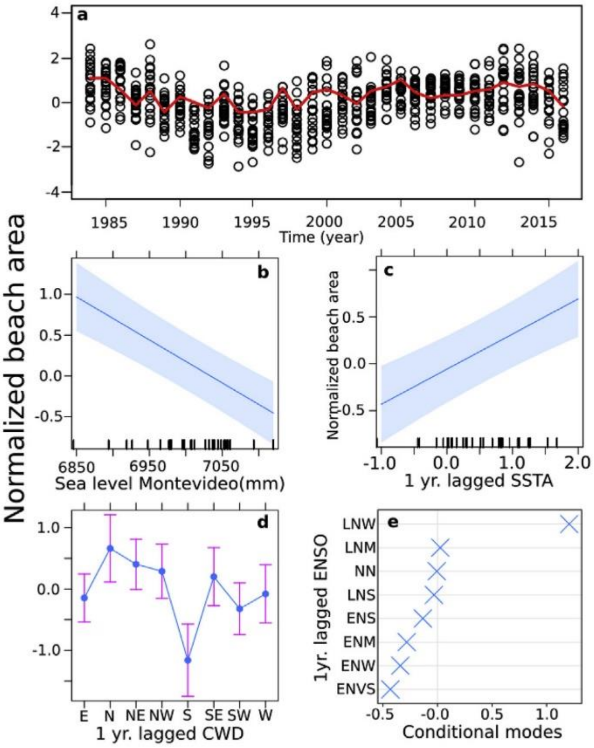

Fig. 3. Normalized beach area variations between 1984 and 2016 and best linear mixed model fitted. a) Scatterplot showing the normalized area for each beach and year, with the continuous red line indicating the best model prediction. Effect plots of: b) Montevideo sea level, c) 1-year lagged sea surface temperature anomaly (SSTA), and d) 1-year lagged categorical wind direction (CWD) on normalized beach area. e) Conditional modes for different ENSO conditions with 1-year lag: La Niña Weak (LNW), El Niño Weak (ENW), Neutral (NN), El Niño Strong (ENS), La Niña Strong (LNS), La Niña Moderate (LNM), El Niño Moderate (ENM) and El Niño Very Strong (ENVS). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

2013). The inverse relationship between sea level and beach area reinforces the notion that the major long-term threat facing sandy beaches worldwide is coastal squeeze (Pontee, 2013), which leaves beaches trapped between rising sea level on the wet side and encroaching development from expanding human populations on land, thus leaving no space for normal sediment dynamics (McLachlan and Defeo, 2018).

Coastal erosion in Montevideo sandy beaches is affecting highly relevant ecological and socioeconomic values (Saizar, 1997; UNESCO, 2010; Gutiérrez et al., 2016). These social-ecological systems (McLachlan and Defeo, 2018) are important for recreation and biodiversity conservation, and therefore should attend a hierarchy of concerns related to public safety, economy and ecology (McLachlan et al., 2013; Elliott et al., 2016). In this context, reconstructing beach area information following the approach developed here could provide insights on ecosystem recovery scenarios and could help outline appropriate management strategies with multiple objectives in these social-ecological systems, particularly in erosive scenarios that require active recovery (Elliott et al., 2007; McLachlan and Defeo, 2018). The restoration of suitable physical conditions will allow system recovery, which in turn will increase ecosystem services and resilience (Balvanera et al., 2014). Furthermore, knowing the climate forcing factors would help predict beach area evolution under different climate change scenarios and also identify key variables for recovery. However, the social-ecological dimensions of beach area loss remain under-explored, as there are no baselines of ecological (e.g., ecosystem services) and economic (e.g., recreation value) indicators for these sandy beaches. Thus, a continued assessment of the effects of climate change should be given a high priority in conservation planning for this coastal region.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2018.12.015.

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5. EFECTO DE LA URBANIZACIÓN SOBRE LA MACROFAUNA EN PLAYAS A LO LARGO DE UN GRADIENTE ESTUARINO

5.1 Resumen

El rápido crecimiento de la población urbana y el desarrollo de infraestructura costera en todo el mundo añaden complejidad y urgencia a los procesos de gestión en costas arenosas. El desarrollo de índices de calidad de playas que tengan en cuenta el papel cada vez más importante de la urbanización es especialmente relevante. Este trabajo analiza el potencial de la densidad poblacional humana (HPD, por su sigla en inglés), las luces nocturnas y la cobertura vegetal como indicadores del impacto urbano en las playas. Cada indicador fue contrastado contra una base de datos de riqueza macrobentónica y características físicas en playas a lo largo de un gradiente ambiental de macroescala, definido por el estuario más ancho del mundo, el Rio de la Plata. La cantidad de luz artificial en la noche alcanzó su punto máximo en la zona más urbanizada de la costa, la ciudad de Montevideo, y mostró una relación negativa con la riqueza macrofaunal. Se encontró una relación lineal negativa entre HPD y el número de especies. La cobertura vegetal, expresada a través del índice de diferencias normalizadas de vegetación, mostró una débil correlación positiva con la riqueza de especies. Regresiones lineales múltiples, comparando la combinación de indicadores urbanos y características físicas como variables explicativas de la riqueza macrofaunal, mostraron los mejores resultados cuando se combinaron HPD y el rango de salinidad. El árbol de regresión explicó el 65% de la desviación y tuvo una estructura coherente con los resultados anteriores. La salinidad fue el forzante ecológico dominante: playas con salinidad ≥ 27.2 mostraron mayor riqueza de especies, mientras que las playas con menor salinidad y mayor iluminación nocturna mostraron la riqueza de especies más baja. Asimismo, el análisis de Random Forest seleccionó la salinidad y HPD como las variables más informativas para discriminar grupos de playas según su riqueza macrobentónica. Estos resultados reflejan que el gradiente de salinidad es un forzante de macroescala que controla los patrones de riqueza de especies a lo largo de esta costa, mientras que los efectos de la urbanización están confinados dentro de éste. La identificación de indicadores urbanos, tales como los proporcionados en este trabajo, constituye el primer paso hacia el desarrollo de enfoques más rigurosos dirigidos a cuantificar la urbanización como factor de estrés duradero y relevante a nivel mundial.

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Urbanization effects on sandy beach macrofauna along an estuarine gradient

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ABSTRACT

The rapid urban population growth and coastal infrastructure development worldwide add dimensionality and complexity to the management process in sandy shores, and therefore the development of beach quality indices that take into account the increasing role of urbanization are particularly relevant. This work analyses the potential of human population density (HPD), nighttime lights and vegetation cover as indicators of urban impact on sandy beach biodiversity. Each indicator was tested against a large-scale 2-yr study of macrofaunal richness and physical characteristics along a strong environmental gradient defined by the widest estuary of the world, the Rio de la Plata. A negative linear relationship between HPD and the number of species was found. The amount of light recorded at night peaked at the most urbanized area in the coast, Montevideo city, and showed a negative relationship with macrofaunal richness. Vegetation cover, expressed through a normalized difference vegetation index, showed a weak positive relationship with species richness. Multiple linear regressions, combining urban indicators and physical characteristics as explanatory variables of macrofaunal richness, showed the best results when HPD and salinity range were combined. A regression tree explained 65% of deviance and had a structure coherent with previous results. Salinity was the dominant ecological driver: beaches with salinity ≥ 27.2 showed higher species richness, while beaches with less salinity and higher nighttime lights showed the lowest species richness. Random Forests selected salinity (mean and range) and HPD as the most informative variables to discriminate groups of beaches according to their macrobenthic richness. These results reflect that the salinity gradient is a macroscale driver that shapes species richness patterns along this coast, whereas the effects of urbanization are confined within the dominant large-scale environmental gradient. The identification of suitable urban indicators provided in this work constitutes the first step onto the development of more rigorous approaches to assess this globally relevant and long-lasting stressor.

1. Introduction

The growth of human populations is considered one of the prime underlying causes of the recent and ongoing decline of species (Woodroffe, 2000). The population density in coastal areas is nearly three times higher than the global average (Small and Nicholls, 2003), with coastlines containing more than two-thirds of the world's largest cities (UN, 2014). Urban areas are characterized by the presence of artificial structures and a high density of people (McDonnell and Pickett, 1990), affecting ecosystems through area loss, disturbance, the spread of invasive species and alteration of biogeochemical cycles (Alberti, 2008). These strongly engineered environments share many similarities despite geographical or climatic differences (Grimm et al., 2008), leading to biodiversity loss through the homogenization of biotic components (Concepción et al., 2015). Coastal structures have significant effects on the ecology of shorelines, especially when entire habitats are replaced with novel materials such as concrete and granite (Todd et al., 2019).

Sandy beaches comprise one-third of the ice-free coasts of the world, providing a wide variety of ecosystem services essential to support human well-being (McLachlan and Defeo, 2018). Beach ecosystems are highly dynamic and primarily defined by the interaction between wave energy, tides and wind regimes (McLachlan et al., 2018). The intertidal areas of beaches provide habitats for a diversity of fauna with unique adaptations to inhabit these harsh systems (Defeo et al., 2009), supplying a wide variety of ecosystem services (McLachlan et al., 2013; Balvanera et al., 2014). The composition of macrobenthic assemblages is mainly controlled by the physical environment, notably the swash climate and water and sand characteristics (Defeo and McLachlan, 2005; Ortega-Cisneros et al., 2011). Given the human preference for coastal areas, urban sandy beaches have become one of the more extended social-ecological systems, and at the same time a

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potential model for assessing the impact of urbanization on ecosystems and its implications on human well-being.

Environmental gradients are important features of ecological systems that challenge our ability to understand and forecast the dynamics of ecological patterns (Bozzeda et al., 2016). Observations of urbanization and ecosystem responses, coupled with large-scale environmental gradients, are crucial for a better understanding of the complex processes currently threatening biodiversity (Grimm et al., 2008). Transitional environments, like estuaries, are useful to estimate how the residing biodiversity changes because of the combined effects of environmental and human-induced drivers. Studies drawn from a pool of widely differing sandy beaches have identified salinity as a key aggregate variable that carries itself different environmental and biotic effects (Atrill, 2002; Ortega-Cisneros et al., 2011; Lercari and Defeo, 2015). This is evidenced in strong estuarine gradients, where the distribution patterns of aquatic organisms are related to salinities influence on critical physiological processes (Atrill, 2002; Lercari and Defeo, 2006; Barboza et al., 2012).

Ecological studies on urban sandy beaches traditionally estimate the level of urbanization through compound indices based on urban indicators such as proximity to urban centers, number of buildings on the sand and beach cleaning intensity (González et al., 2014). These indicators are usually systematized into low, medium, and high levels of development and estimated by direct observations in the field or available data from local managers (Cardoso et al., 2016; Machado et al., 2016; Costa et al., 2017; Laitano et al., 2019). This approach has provided insights on the effects of urbanization on sandy beach macrofauna (Cardoso et al., 2016; Costa et al., 2017), insects (González et al., 2014) and the resilience of the macrobenthic community to this perturbation source (Machado et al., 2016; Laitano et al., 2019). However, scientists' values, interests and background bias the choice of which components and interactions are needed to understand the relationships between urbanization and the ecological characteristics of beaches (Bombana and Ariza, 2018). The value of an ecological indicator is inversely related to the uncertainty of its estimation, and categorizing conditions through perception-based estimations introduces uncertainty, limiting the reliability of results (Carstensen and Lindegarth, 2016). This approach hampers the scale and extent of pattern analysis and generalizations on a globally relevant matter (Hahs and McDonnell, 2006; McLachlan et al., 2018).

A stronger characterization of biodiversity spatial patterns is a critical challenge for understanding the effects of urbanization on marine ecosystems and processes (Todd et al., 2019). In this paper, urbanization, macrofaunal richness and physical variables at 16 sandy beaches located along a macroscale estuarine gradient are analyzed with the objectives of: (1) assessing the concurrent impacts of environmental and urbanization variables on macrobenthic richness; and (2) exploring the suitability of vegetation cover, human population density and nighttime lights as objective urbanization indicators, and rank them based on its importance over sandy beach macrofaunal richness.

2. Material and methods

2.1. Study area

The Rio de la Plata estuary is located on the west coast of the South Atlantic Ocean (Fig. 1). Landward waters from Parana and Uruguay rivers drain an annual average discharge of 22000 $m^3 \cdot s^{-1}$ (Simionato et al., 2001) forming a shallow (depths up to 15 m) coastal-plain estuary. A turbidity front located around Montevideo city (beaches 3 and 4, Fig. 1a) constitutes the surface indication of the transition between fresh and saline waters (Sepúlveda et al., 2004). The Uruguayan coastline stretches over 320 km along the northeastern bank of the Rio de la Plata estuary and 210 km of adjacent Atlantic coastline (Lercari and Defeo, 2006). This coast is characterized by sandy beaches interrupted by rocky heads, with a semidiurnal microtidal regime (ca. 0.5 m of amplitude) (Lercari and Defeo, 2006). Sandy beaches distributed along the gradient of the Rio de la Plata, the widest estuary worldwide, provide an ideal framework for evaluating the concurrent effects of the environment and urbanization over species richness. At this coast, the large-scale spatial distribution of macrofaunal sandy beach communities has been related to salinity range and, to a minor extent, to the width of the swash zone (Lercari and Defeo, 2006, 2015).

2.2. Biological and physical information

The macrofauna and environmental characteristics of 16 sandy beaches on the coast of Uruguay (Fig. 1a) were sampled bimonthly from June 1999 to May 2001. Environmental variables recorded were beach slope and width, swash zone width, water salinity and temperature, and sediment granulometry (methodological details in Lercari and Defeo, 2006). Three transects were placed perpendicularly to the shoreline and spaced 8 m apart, with sampling units (SUs) starting from the base of the dunes and continuing at 4-m intervals in a seaward direction until reaching the lower limit of the swash zone. SUs at each transect were taken with a metal cylinder (diameter: 27 cm, length: 40 cm); macrofaunal samples were sieved through a 0.5-mm mesh and the retained macrofauna was fixed (formalin 5%) for identification (Lercari and Defeo, 2015). The salinity range was measured as the difference between lowest and highest salinity values during the study period, giving a proxy of the stress imposed by salinity variations at each location (Atrill, 2002; Lercari and Defeo, 2006).

2.3. Urbanization

Three of the most widely used indicators of urbanization are human population density (Woodroffe, 2000; McKinney, 2002), vegetation cover (Morawitz et al., 2006; Alberti, 2008) and nighttime light intensity (Small and Nicholls, 2003; Zhou et al., 2014). To estimate urban development along the large-scale estuarine coast of Uruguay, the Google Earth Engine geographic information platform (Gorelick et al., 2017) was applied to extract information on the above-mentioned indicators from openly available global databases. Satellite information has been highlighted as the most realistic way forward in providing useful data for studies of nearshore dynamics (Short and Jackson, 2013) and has been used for estimation of beach characteristics (Harris et al., 2011) and coastal monitoring (Orlando et al., 2019). Polygons enclosing each of the study sites were constructed based on Quickbird satellite images. The landward limit of the beach was set on areas where vegetation covered sand, or when a road was present. To represent the degree of urbanization around the beach, all urban variables were extracted as mean values from the shoreline in a landward circular area that excludes the beach polygon itself (Fig. 1b). To evaluate different areas of influence for each urban variable, and to help elucidating the relationship between sandy beaches and their neighboring land, radiuses of 500, 1000, 2000 and 3000 m were considered.

Nighttime light intensity (NL) was estimated as the amount of light observed at night to map the extent and dynamics of urban areas (Zhou et al., 2014). This variable is related to economic activity and urban infrastructure and has been used to estimate global urbanization trends (Small and Nicholls, 2003). NL was estimated through the stable nighttime lights band of the Defense Meteorological Satellite Program/Operational Linescan System, using yearly averages of NL in an integer scale of 1 to 63, over a 1-km grid. Data from years 2000 and 2001 were combined to better represent the studied period. Vegetation cover can act as an inverse urbanization index, with higher values indicating greener areas, and therefore less urbanization (Alberti, 2008). This indicator was estimated through the normalized difference vegetation index (NDVI), which distinguishes between vegetation and non-vegetation cover (Morawitz et al., 2006). NDVI was calculated on a cloud-free Landsat 5 composite of years 2000 and 2001, as a ratio of near-infrared (NIR) and red bands (NDVI = (NIR − Red)/(NIR + Red).

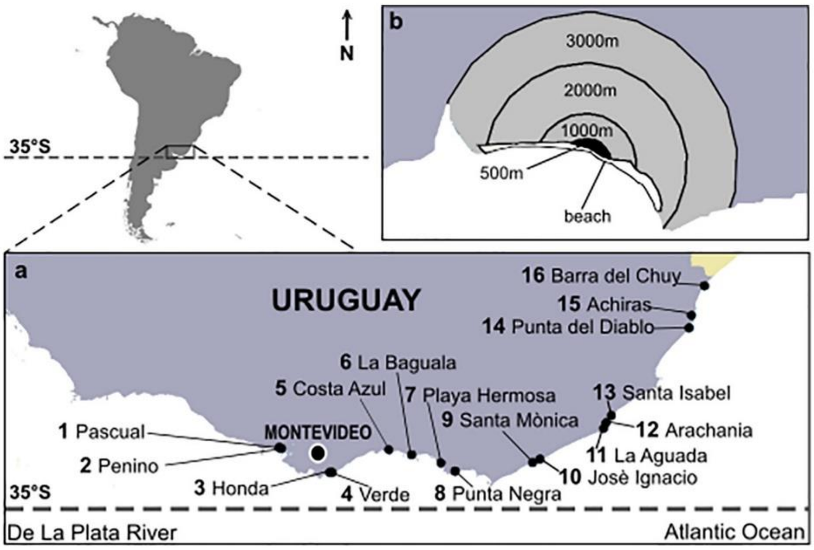

Fig. 1. Study area, studied beaches and levels of urban variables acquisition. a) Beaches from west to east (distance from westernmost site): 1 Pascual (km 0); 2 Penino (km 2); 3 Honda (km 51); 4 Verde (km 52); 5 Costa Azul (km 108); 6 La Baguala (km 112); 7 Playa Hermosa (km 125); 8 Playa Hermosa (km 139); 9 Santa Mónica (km 197); 10 José Ignacio (km 203); 11 La Aguada (km 257); 12 Arachania (km 260); 13 Santa Isabel (km 267); 14 Punta del Diablo (km 350); 15 Achiras (km 356); 16 Barra del Chuy (km 378). The black dot with a white outline indicates the capital city, Montevideo. b) Example of areas of urban variables data acquisition; beach area is the white area enclosed in a polygon, the black area shows the 500 m radius and grey circles show the 1000 to 3000 m radiuses from the beach centroid.

Human population density (HPD) was estimated through the Gridded Population of World version 4 (CIESIN, 2016). This database, models the distribution of the global human population, based on official census information, on a grid of 1-km cells. In this case, the information is based on the 2004 Uruguayan national census (INE, 2004). HPD is widely used as an urbanization proxy and has been proven relevant to test hypotheses about the relationships between urban and ecological functions along urban to rural gradients (Woodroffe, 2000; McKinney, 2002; Alberti, 2008). Road density was initially considered as one of the proxies for urbanization due to the extensive literature on this subject (Coffin, 2007). However, it was impossible to acquire reliable information for the study period.

2.4. Data analysis

Simple linear modelling was chosen to explore the relationship between species richness and urban and physical variables (Bolker et al., 2009). All urban variables were considered on the four measured levels (500, 1000, 2000 and 3000 m radiuses), and HPD was log-transformed to fit distribution requirements. The residuals of the linear relationship between species richness and salinity range were used to explore their relationship with all non-salinity variables, once the effect of salinity variability on species richness was removed (Zuur et al., 2009). Physical variables that showed a significant correlation with macrobenthic richness were considered to assess the combined effects of urbanization and beach characteristics. To this end, multiple linear models were fitted using urban and physical indicators. Due to the number of sampling sites (16), models were limited to two independent variables to ensure enough degrees of freedom for an accurate estimation of regression coefficients (Zuur et al., 2009). Multiple linear models covering all possible variable combinations were constructed and compared by the Akaike Information Criteria corrected for small samples (AICc) (Hurvich and Tsai, 1989). Regression trees, considering all variables, were built to identify key ecological and urban attributes, and to determine the way in which these variables would influence species richness (De'ath and Fabricius, 2000). The regression tree algorithm in *rpart* package of R Software (R Core Team, 2013) was used to build trees by iteratively partitioning the dataset into a nested series of mutually exclusive groups. The final tree was obtained by generating a large tree that was pruned afterwards to reduce overfitting (Gutiérrez et al., 2011). To evaluate the importance of each variable and give further support to the regression tree results, a Random Forest with 10,000 regression trees, each using two randomly selected variables, was built using the R package *randomForest* (Prasad et al., 2006).

3. Results

Salinity (mean and range) and swash width were the only physical variables that showed a significant relationship with macrobenthic richness (Supplementary materials A). Both salinity indicators showed a pattern coherent with the influence of the freshwater input from the inner estuary (Fig. 2). Mean salinity increased from beach 1 to 16 following the main estuary axis (Fig. 2a). The salinity range showed a humped pattern, with the highest values found in mid-estuary areas (Fig. 2b). Species richness increased with mean salinity (Fig. 2d), decreased with salinity range (Fig. 2e) and increased with swash width (Fig. 2f).

Western beaches showed higher human presence and nighttime light activity, and lower vegetation cover (Fig. 3). HPD peaked in Montevideo city (beaches 3 and 4), with eastern local peaks at beaches 11 and 14 being associated with year-round stable economic activities (Fig. 3a). A strong negative linear relationship between HPD and the number of species was found (Fig. 3d), presenting the highest R^2 and the lowest AICc value of all regressions (Supplementary materials B). NDVI showed little variation among beaches, with the lowest value occurring in Montevideo city and the highest at beach 15 (Fig. 3b). A weak positive relationship between NDVI and macrofaunal species richness was found (Fig. 3e). NL peaked in Montevideo city (Fig. 3c) and showed a negative relationship with macrofaunal richness (Fig. 3f). The residuals between species richness and salinity range were negatively correlated with HPD in a 2000 m radius ($R^2 = 0.32$, $p < 0.05$) (Supplementary materials C), showing that HPD has significant correlation with species richness when the effect of salinity is removed from the analysis.

Multiple linear regressions showed the best results when HPD and salinity range were jointly used as predictor variables (Table 1). However, according to AICc, there were no relevant differences between models with different HPD levels and a model using swash width and salinity range. Models including NL and NDVI (3000 m radius) also showed a good explanatory power when combined with salinity range; other models including NDVI were non-significant (Table 1).

The regression tree explained 65% of deviance and had a structure coherent with the results shown previously (Fig. 4a). Mean salinity was the dominant variable: beaches with salinities ≥ 27.2 showed a mean

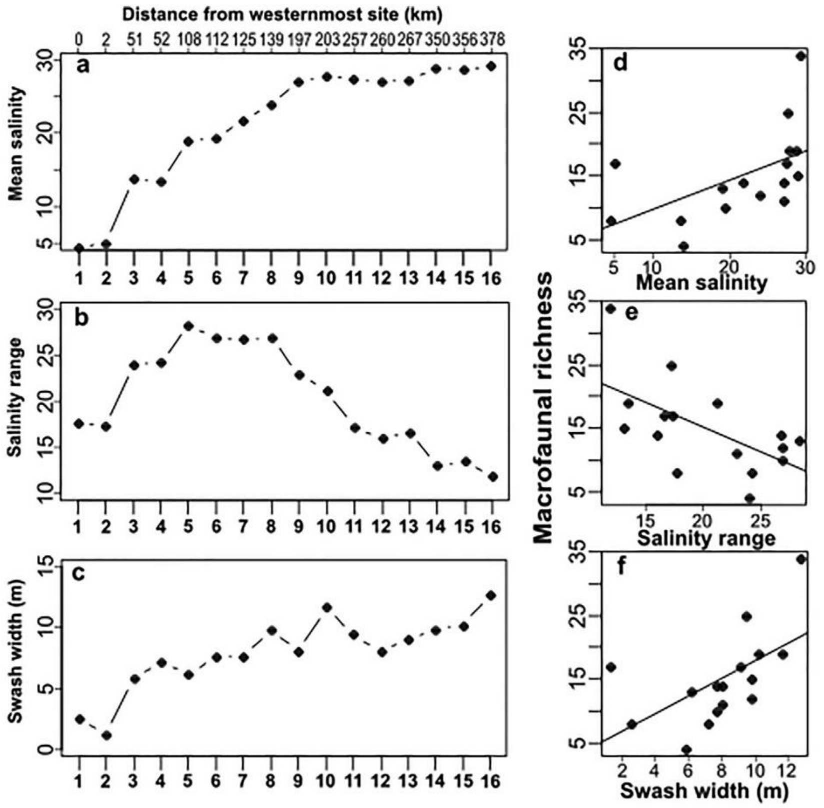

Fig. 2. Large-scale distribution of selected physical variables in 16 Uruguayan sandy beaches: a) mean salinity, b) salinity range, calculated as the difference between maximum and minimum salinity, c) swash width in meters. Linear relationships between macrofaunal species richness and d) mean salinity (R^2 = 0.28, $p < 0.05$), e) salinity range (R^2 = 0.36, $p < 0.05$) and f) swash width (R^2 = 0.33, $p < 0.05$). Beaches are numbered from west to east following Fig. 1.

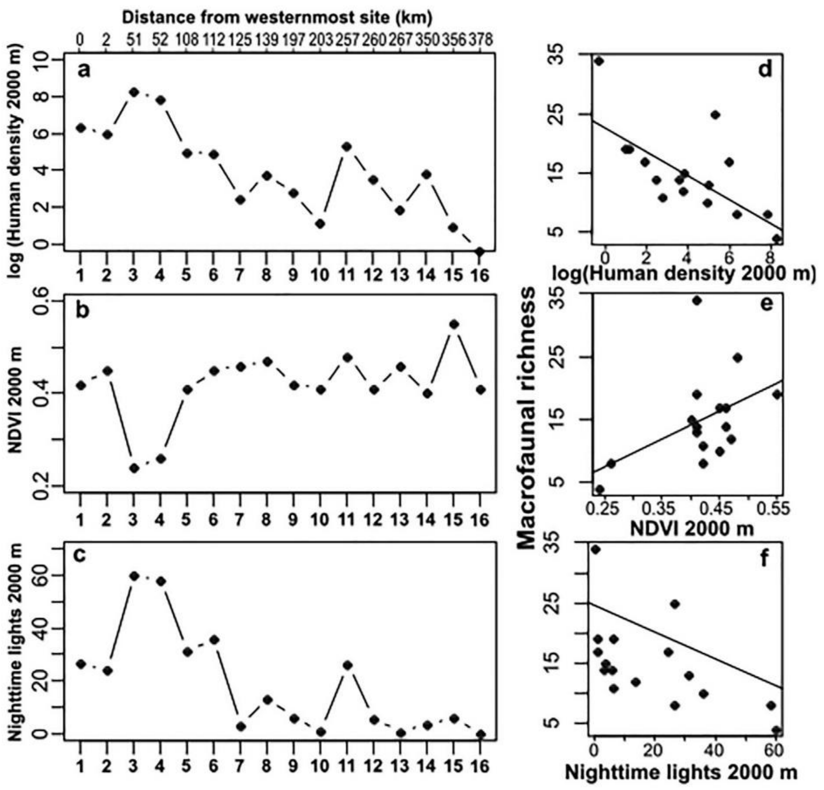

Fig. 3. Large-scale distribution of selected urban variables measured on a 2000 m radius at 16 Uruguayan sandy beaches: a) logarithm of human density, b) normalized difference vegetation index (NDVI), c) stable nighttime lights. Linear relationships between macrofaunal species richness and d) logarithm of human density (R^2 = 0.47, $p < 0.05$), e) NDVI (R^2 = 0.22, $p < 0.1$) and f) stable nighttime lights (R^2 = 0.33, $p < 0.05$). Beaches are numbered from west to east (distance from the westernmost site) following Fig. 1.

Table 1
Summary of the best 10 linear models relating macrofaunal richness to urban and ecological variables. The column 'Variables' details the predictive variables for species richness in linear models. All listed models have 4 degrees of freedom. Urban variables are measured in a diameter expressed along with the name: logarithm of human population density (HPD), normalized difference vegetation index (NDVI) and stable nighttime lights (NL). AICc is the Akaike's information criteria for small samples; R^2 is the multiple coefficient of determination and Adj. R^2 is the adjusted coefficient of determination. Only models with $p < 0.05$ are shown.

Variables	AICc	R^2	Adj. R^2
HPD (2000 m) + Salinity range	104.3	0.60	0.54
HPD (3000 m) + Salinity range	104.9	0.59	0.52
HPD (1000 m) + Salinity range	105.1	0.58	0.52
Salinity range + Swash width	105.9	0.56	0.49
HPD (500 m) + Salinity range	106.5	0.54	0.47
NL (3000 m) + Salinity range	108.2	0.49	0.42
NDVI (3000 m) + Salinity range	108.2	0.49	0.42
HPD (2000 m) + Swash width	108.5	0.49	0.41
NL (1000 m) + Salinity range	108.7	0.48	0.40
NL (2000 m) + Salinity range	108.7	0.48	0.40

species richness of 22. Beaches with salinities < 27.2 were subdivided according to NL (2000 m radius): those with ≥25.3 units of incident NL had lower average species richness (9) than those with less NL in the vicinity (14). The Random Forest procedure selected salinity (mean and range) as the most informative variables to discriminate beach groups according to their species richness (Fig. 4b). The third variable in importance was HPD (1000 m radius), followed by NL (2000 m). NDVI showed the least importance, which could be attributed to the relatively uniform values obtained for this variable.

4. Discussion

This study reinforces the role of urbanization in explaining biodiversity patterns in coastal ecosystems, a greatly needed input to the nascent field of urban marine ecology given the current global context (Todd et al., 2019). The large-scale analysis of sandy beach macrofauna showed that sites with more urbanized surrounding areas had less macrofaunal richness, whether the considered indicator was, in decreasing order of relevance, nighttime lights, human density and vegetation cover. Particularly, strong negative correlations were found between the urban indicators HPD and NL, and sandy beach macrofaunal richness, and these results were confirmed by regression trees and Random Forest analysis. However, the importance of these indicators was constrained within the effect of a large-scale environmental salinity gradient generated by the Rio de la Plata, highlighting the relevance of considering environmental gradients to further understand the relationships acting on these complex physically-driven systems (Lercari and Defeo, 2006, 2015; Barboza et al., 2012).

The human occupation at the Uruguayan coast is primarily characterized by low-density seasonal tourism. Urban development is generally associated with key locations like industrial or artisanal fishing ports, and suburban sprawl occurs profiting the services associated with these locations and the economic impulse of seasonal tourism. This pattern has exceptions, such as Montevideo city (beaches 3 and 4), where coastal occupancy is high all year-round, and some very urbanized high-end touristic sites not included in this study (e.g., Punta del Este). To foster urban development, a major substitution of tree cover was implemented at the beginning of the last century along the Uruguayan coast. *Acacia longifolia* and *Pinus* spp. were planted to stabilize the dunes, leaving native coastal forests confined to relictual areas (Delfino et al., 2011). The weak positive relationship between vegetation cover and macrobenthic richness can be ascribed to non-native vegetation having high NDVI values (Morawitz et al., 2006). It is expected that species substitution associated with urbanization harms biodiversity (Alberti, 2008).

Light pollution is an anthropogenic threat for biodiversity in general (Gaston et al., 2013), and sandy beach organisms are no exception (Luarte et al., 2016; Manríquez et al., 2019). Nighttime lights are likely to produce important negative effects on biological processes, affecting species mainly through the physiological and behavioral consequences of altering natural light cycles (Gaston et al., 2013; Manríquez et al., 2019; Scapini et al., 2019). A direct negative impact of NL has been documented for sandy beach macrofaunal components, such as insects (González et al., 2014), amphipods (Luarte et al., 2016) and gastropods (Manríquez et al., 2019). The NL variable employed in this study is based on annual means and therefore captures the increased energetic expenditure associated with summer visitors.

Increasing numbers of residents and tourists have negative ecological effects, including trampling, which is emerging as a significant environmental issue in urbanized sandy shores, because it can (reviewed in McLachlan and Defeo, 2018): (1) decrease sand stability by changing compaction and increasing its mobility, and change the organic matter and moisture content of the soil; (2) decrease biomass, individual growth and survival of beach plants and cause disappearance

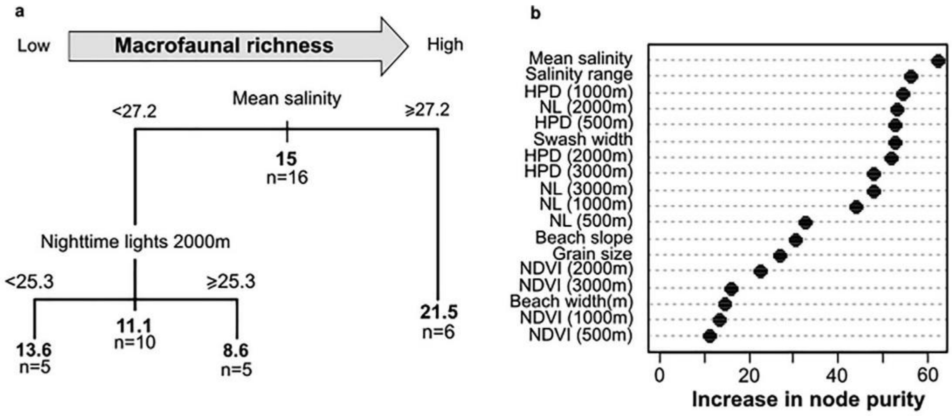

Fig. 4. Graphical representations of classification procedures. a) Regression tree showing the hierarchical relationship between the most important variables explaining 65% of the total deviance. Splitting criteria are shown above each node, with the bold number below each node being the average value for macrofaunal richness. n is the number of sandy beaches at each node. b) The importance of variables determined by Random Forests as purity increase at each split: the value shown for each variable is the difference between the residual sum of squares before and after the split, averaged over all trees. Urban variables are measured in a diameter expressed along with the name: logarithm of human population density (HPD), normalized difference vegetation index (NDVI), stable nighttime lights (NL).

of vulnerable species; and (3) disrupt fauna, affecting species richness and density and altering assemblage structure up to the point of producing a significant simplification of the community (Reyes-Martínez et al., 2015). The effect of trampling is aggravated by concurrent habitat modifications typically performed in urbanized sites, such as seawalls replacing foredunes, mechanical beach cleaning, and alteration of freshwater discharges. Under such circumstances, beach habitats could become unsuitable for sandy beach macrofauna (Defeo et al., 2009; McLachlan and Defeo, 2018; Jorge-Romero et al., 2019). These ecosystems are important for human well-being through recreation services, biodiversity conservation, increased storm protection, and therefore their management should attend a hierarchy of concerns related to public safety, economy and ecology (Elliott et al., 2016). Resolving, or minimizing, conflicts between people and wildlife within the urban context is vital to avert local extirpations, and central to the management of urban ecosystems.

The salinity gradient was identified by regression trees and Random Forests as the main explanatory driver of the macroscale distribution of the macrofauna along the Uruguayan coast (Fig. 3, see also Lercari and Defeo, 2006). Salinity is a macroscale driver with an evolutionary time of action, shaping the species richness patterns along this coast by affecting the distribution of aquatic organisms and influencing critical physiological processes. The width of the swash zone, an acknowledged driver of macrobenthic diversity correlated with food and habitat availability for intertidal organisms (McLachlan and Defeo, 2018), was outperformed by NL in all analysis. As both variables operate locally, the results suggest that urbanization could affect the effective habitat availability. This is coincident with studies conducted on other urban ecosystems (Alberti, 2008) and has major implications for macroecological analysis of sandy beach diversity.

Studies covering a larger geographic and environmental range are required to further advance in this long-postponed matter. This scale of analysis can be achieved combining the globally available urban indicators analyzed here and satellite-based estimation of ecological features of sandy beaches (Harris et al., 2011; Orlando et al., 2019). Seasonal touristic beaches, such as several included in this study, pose a challenge and an opportunity to this scale of analysis: infrastructure is constant throughout year, but direct human impact is higher during the summer season (Laitano et al., 2019). This constitutes a *de facto* experiment that could help elucidate the relationships and implications of different anthropogenic drivers over sandy beach ecosystems. Finally, it is important to note that the quantification of urbanization impact on sandy beaches does not seek to deny the use of perception-based indices, but rather to provide a complementary, more rigorous, tool. The rapid urban population growth and coastal infrastructure development worldwide add dimensionality and complexity to the management process in sandy shores, and therefore the development of beach quality indices directed to quantify the increasing role of urbanization is particularly relevant. The identification of suitable urban indicators provided in this work constitutes the first step onto the development of more rigorous approaches to assess the impact of this relevant and long-lasting stressor on sandy beach ecosystems.

CRediT authorship contribution statement

L. Orlando: Conceptualization, Methodology, Software, Visualization, Writing - original draft. **L. Ortega:** Conceptualization, Formal analysis, Supervision, Writing - review & editing. **O. Defeo:** Conceptualization, Investigation, Supervision, Writing - review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2019.106036.

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5.3 Material Suplementario

Urbanization effects on sandy beach macrofauna along an estuarine gradient

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Supplementary material

Section A

Supplementary Table. Summary of the linear models relating mean number of species to physical variables. df are the degrees of freedom, AICc is the Akaike's information criteria for small samples, R^2 is the multiple coefficient of determination, and Adj. R^2 is the adjusted one. All models are statistical significant ($p<0.05$).

Variables	df	AICc	R^2	Adj. R^2
Salinity Range	3	108.25	0.36	0.32
Mean Salinity	3	110.15	0.28	0.23
Swash width	3	109.17	0.33	0.28

Section B

Supplementary Table. Summary of the linear models relating mean number of species to urbanization. All urban variables were extracted as mean values from the shoreline in a landward circular area and the diameter is expressed along the name. HPD is the human population density, log-scaled; NDVI is the normalized difference vegetation index, NL is the stable nighttime lights, df are the degrees of freedom, AICc is the Akaike's information criteria for small samples, R^2 is the multiple coefficient of determination and Adj. R^2 is the adjusted one. All models are statistically significant ($p<0.05$), with the exception of * $p<0.1$.

Variables	df	AICc	R^2	Adj. R^2
HPD (2000m)	3	105.3004	0.47	0.43
HPD (1000m)	3	105.9018	0.45	0.41
HPD (3000m)	3	106.5711	0.43	0.39
HPD (500m)	3	106.7315	0.42	0.38
NL (3000m)	3	108.5283	0.35	0.31
NL (1000m)	3	108.9976	0.33	0.29
NL (2000m)	3	109.0425	0.33	0.28
NL (500m)	3	109.353	0.32	0.27
NDVI (2000m)*	3	111.6031	0.22	0.16

Section C

Supplementary Table. Summary of the linear models relating the residuals of the best model fitted in Section A (salinity range) with all non-salinity variables. All urban variables were extracted as mean values from the shoreline in a landward circular area and the diameter is expressed along the name. HPD is the human population density, log-scaled; NDVI is the normalized difference vegetation index, NL is the stable nighttime lights, df are the degrees of freedom, AICc is the Akaike's information criteria for small samples, R^2 is the multiple coefficient of determination and Adj. R^2 is the adjusted one, and p-value reports the significance of each model.

Variables	AICc	R^2	Adj. R^2	p-value
HPD (2000m)	100.1	0.32	0.27	0.023
HPD (3000m)	100.4	0.31	0.26	0.026
Swash width	100.7	0.29	0.24	0.030
HPD (1000m)	100.8	0.29	0.24	0.032
HPD (500m)	102.2	0.23	0.17	0.063
NDVI (3000m)	102.6	0.20	0.15	0.080
NL (3000m)	103.4	0.16	0.10	0.123
Beach slope	103.5	0.16	0.10	0.126
NDVI (2000m)	103.7	0.15	0.09	0.138
NL (1000m)	103.8	0.14	0.08	0.152
NL (2000m)	103.8	0.14	0.08	0.153
Grain size	104.0	0.13	0.07	0.164
NL (500m)	104.1	0.13	0.07	0.174
NDVI (1000m)	105.8	0.03	-0.04	0.534
Beach width	105.9	0.02	-0.05	0.563
NDVI (500m)	106.3	0.00	-0.07	0.999

6. PERSPECTIVAS PARA EL MANEJO DE PLAYAS EN EL ANTROPOCENO: INFORMACIÓN SATELITAL, ESTACIONALIDAD TURÍSTICA Y OPINIÓN DE EXPERTOS

6.1 Resumen

Las playas de arena combinan relevancia ecológica con un potencial de desarrollo humano ampliamente reconocido. La gestión de estos ecosistemas es una tarea compleja que se basa principalmente en herramientas de clasificación que dependen de percepciones de expertos. Este enfoque no abarca toda la complejidad de estos sistemas, limitando la escala, la calidad de los análisis comparativos y la detección de patrones. Este trabajo incorporó indicadores cuantitativos disponibles a nivel mundial para la gestión de playas arenosas mediante el modelado de las percepciones de los expertos sobre el potencial de conservación y recreación para 58 playas en cuatro continentes. La estacionalidad turística se estimó a través de indicadores de pulso con base en las variaciones mensuales de las luces artificiales nocturnas (ALAN, por su sigla en inglés) y se contrastó su relación con la temperatura del agua y el grado de urbanización en cada sitio. Se modelaron indicadores de estacionalidad, atributos de los ecosistemas y un conjunto de descriptores de urbanización derivados de información satelital, para predecir puntajes derivados de la aplicación de dos índices de playa desarrollados para evaluar el potencial de conservación y recreación, basados en la opinión de expertos. El análisis de patrón envolvente mostró una relación en forma de campana entre indicadores de turismo estacional y la temperatura superficial del mar, así como una tendencia decreciente con la urbanización. Un árbol de regresión explicó el 73% de la desvianza del potencial de conservación y mostró una estructura coherente con las consideraciones del índice sobre el estado de las dunas, las especies icónicas y la riqueza macrobentónica. El análisis de Random Forest explicó el 62% de la varianza, y ALAN fue la variable más significativamente asociada con las prioridades de conservación adjudicadas por expertos, seguida de la densidad humana y las áreas de playa y arena. El árbol de regresión explicó el 49% de la desvianza en la recreación dependiendo principalmente de la magnitud del turismo estacional y las características de la urbanización. El análisis de Random Forest explicó el 23% de la varianza, principalmente dada por la magnitud del turismo estacional y ALAN, mientras que la cobertura vegetal y el área de arena también fueron relevantes. Los resultados sugieren un alto potencial para la incorporación de información satelital de acceso abierto al manejo de playas arenosas, particularmente aquellas variables que describen actividad humana. ALAN y la magnitud del turismo estacional, ambos antropogénicos, mostraron el mejor desempeño como variables cuantificadas que explican la opinión de los

expertos en cuanto a conservación y recreación. Un enfoque combinado y complementario que combine la información científica derivada de la información basada en satélites y el conocimiento de expertos podría conducir a una aproximación más objetiva sobre el uso potencial de las playas para recreación o conservación. La integración de estos diferentes formatos de conocimiento podría aumentar la capacidad de explorar, promover, fortalecer y apoyar estrategias de gestión para mejorar el estado de estos ecosistemas frágiles y en riesgo creciente.

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Perspectives for sandy beach management in the Anthropocene: Satellite information, tourism seasonality, and expert recommendations

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ABSTRACT

Sandy beaches combine a highly documented ecological relevance with a widely acknowledged human development potential. Management of these mainly physically-driven ecosystems is a complex task that relies primarily on ranking protocols for site assessment, which is a procedure dependent on expert perceptions. This approach fails to address the systems' complexity, hampering the scale and quality of comparative analysis and pattern detection. This work evaluates the incorporation of globally available quantitative indicators to sandy beach management by modeling expert perceptions on conservation and recreation potential for 58 beaches in four continents. Tourism seasonality was estimated through pulse indicators based on monthly variations of Artificial Lights At Night (ALAN) and then contrasted with water temperature and urbanization. Seasonality indicators, ecosystem attributes, and a set of urbanization descriptors derived from satellite information were modeled to predict expert-based scores derived from two beach indices developed to assess conservation and recreation potential. A constrained envelope analysis showed a bell-shaped pattern of seasonal tourism indicators with sea surface temperature and a decreasing trend with ALAN. A regression tree explained 73% of conservation potential deviance and displayed a structure coherent with the index considerations on dune state, iconic species, and macrobenthic richness. Random Forest analysis explained 62% of the variance, and ALAN was the variable most significantly associated with expert advised conservation priorities, followed by human density and beach and sand areas. The regression tree explained 49% of deviance on recreation and depended on seasonal tourism magnitude and urbanization features. The Random Forest analysis explained 23% of the variance, mainly given by seasonal tourism magnitude and ALAN; vegetation cover and sand area were also relevant. The results suggest a high potential in incorporating freely distributed satellite information in sandy beach management, particularly those describing human patterns. ALAN and seasonal tourism magnitude, both anthropogenic, best-ranked as measurable variables explaining expert opinion on conservation and recreation. A combined and complementary approach merging scientific information derived from satellite-based information and expert knowledge could lead to a coherent management narrative about the potential use of beaches for recreation and or conservation. Integrating these different knowledge formats could increase the ability to explore, promote, strengthen and support management strategies to improve the status of a fragile ecosystem at risk increasingly threatened by several stressors acting together.

1. Introduction

Sandy beaches are critical for human well-being in coastal areas, combining a highly documented ecological relevance (McLachlan and Defeo, 2018) with a widely acknowledged human development potential (Stathakis and Baltas, 2018; Fanini et al., 2020). However, these dynamic ecosystems (Short, 1999; McLachlan et al., 2018) and the services they provide are under increasing stress due to the escalating deleterious effects of several stressors acting together, notably including climate change (Barnard et al., 2015; Orlando et al., 2019), increasing use of resources and exacerbated urbanization and industrial development (Small and Nicholls, 2003; Seto et al., 2012). This "triple whammy" effect (Defeo and Elliott, 2021) has led to extensive disruption and ecosystem fragmentation and replacement (Reyes-Martínez et al., 2015; Defeo et al., 2021). Mainly, the rising numbers of residents and tourists impose increasing pressure on beaches, emerging as a

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significant environmental issue on urbanized shores (Orlando et al., 2020). In addition, the high recreational potential makes sandy beaches perceived as sea and sun spaces, facing a lack of recognition as ecosystems vulnerable to a suite of stressors and an absence of alignment in management decisions (Fanini et al., 2020; Defeo et al., 2021).

Ecosystem management requires understanding the interacting, interrelated, and interdependent sub-systems comprising ecological, societal and management complexity (Elliott et al., 2020). Sandy beach management has been addressed relying on ranking tools strongly dependent on expert perceptions (Krueger et al., 2012), partly because on-site measures present a high cost and logistic complexity (McLachlan et al., 2013; Short and Jackson, 2013). Expert-based sandy beach management approaches are valuable and valid; however, they fail to address the existing complexity behind these ecosystems (Bombana and Ariza, 2018), thus hampering the scale and quality of comparative analysis and pattern detection (Carstensen and Lindegarth, 2016). Resolving trade-offs between conservation and recreation is vital to avert degradation and central to managing these ecosystems in the Anthropocene (Orlando et al., 2020).

Given the current and rising pressure over sandy beaches, incorporating reliable and objective SMART (Specific, Measurable, Achievable, Realistic, Time-bounded) indicators into beach management is a pressing need (Cormier and Elliott, 2017). In this context, satellite-based estimations of ecosystem characteristics provide low-cost, standardized proxies of features relevant to sandy beach management, such as coastline variations (Vos et al., 2019), land cover (Orlando et al., 2020), and morphodynamic state (Harris et al., 2011). However, management of these social-ecological systems (sensu Ostrom, 2009) also requires components that describe human use dynamics (Fanini et al., 2020), which has been previously addressed through perception-based indices (González et al., 2014; Laitano et al., 2019; McLachlan et al., 2013) and human occupancy data (Santos et al., 2019). Concerning the latter, the increasing availability of satellite-based information on Artificial Lights at Night (ALAN) could provide unique information to estimate variations in human use (Stathakis and Baltas, 2018). Seasonal beach use poses an additional challenge to management (Ariza et al., 2008; Laitano et al., 2019): infrastructure is constant throughout the year, acting as a press disturbance that can produce long-term changes in the impact trajectory (Harris et al., 2018), whereas direct human impact is higher during the summer season, functioning as a pulse disturbance that can trigger changes in ecosystem state (Jentsch and White, 2019). Moreover, incorporating seasonal beach use in a standardized manner could improve management capacities (Fanini et al., 2020) and could be helpful to assess the cumulative impacts of multiple human activities, which is an increasingly important and complex management issue of the marine environment (Lonsdale et al., 2020).

The increasing availability of satellite information allows quantification and incorporating ecological and socioeconomic SMART indicators into management frameworks. This work evaluates the current potential of improving sandy beach management by including globally available quantitative indicators. An objective and rigorous perspective on sandy beach optimal use is provided by contrasting measurable indicators with expert recommendations. Tourism seasonality, ecosystem attributes and a set of urbanization indicators derived from satellite information were modeled through regression trees to predict scores from two widely used beach indices developed by McLachlan et al. (2013) to assess conservation and recreation potential. A Random Forest approach was also used to rank each variable importance as an indicator of conservation and recreation, giving further insights into the relationships between use recommendations and ecosystem characteristics.

2. Methods

Globally available quantitative indicators of sandy beach features were used for modeling conservation and recreation potential to provide additional insights on sandy beach management. To this end, ecosystem attributes, urbanization, and seasonal tourism were quantified for all sites; a beach polygon was constructed for ecosystem delimitation. In addition, an adjacent land polygon was defined to measure variables associated with the beach vicinity. The landward limit of beach polygons was established by either hard structures or vegetation entirely covered the sand, depending on the degree of coastal armoring (Orlando et al., 2019). The land polygon was constructed as a 2000 m landward expansion of the beach polygon (Supplementary materials A).

2.1. Expert recommendations

The reference of expert opinion was obtained from the results of two expert-based indices developed by McLachlan et al. (2013) (Table 1): 1) the conservation index (CI), which is based on the health state of dunes, the presence of iconic species and macrobenthic species richness; and 2) the recreation index (RI), based on the extent of infrastructure, the level of safety/health of a beach, and its physical carrying capacity. This method has been extensively applied, providing a reference for comparison with the estimates obtained here based on satellite information. To this end, a database of studies providing full information on all scores that compose the index was constructed, covering published information for 58 beaches from 6 countries: Australia (7), Brazil (22), Chile (13), Uruguay (7), South Africa (3), and Spain (6) (Fig. 1; Supplementary materials B and E) (Cardoso et al., 2016; González and Holtmann-Ahumada, 2017; McLachlan et al., 2013; Muñoz-Lechuga et al., 2018).

2.2. Ecosystem attributes

The following sandy beach attributes were estimated inside the beach polygon: sand area, vegetation cover, and beach area. To separate sand from other kinds of cover (e.g. rocky, hard structures, vegetation), a Random Forest classification was applied to an annual composite image constructed with non-cloudy pixels from the Landsat 7 collection. Training areas for sand and other covers were selected for each beach, and 100 training points were randomly extracted to train the algorithm. Sand area was estimated as the area of pixels classified as sand inside the beach polygon (Orlando et al., 2019). Beach pixels that were not classified as sand and had a Normalized Differences Vegetation Index (NDVI) greater than 0.25 were used to determine beach vegetation area (Marzialetti et al., 2019). The total beach area was obtained as the sum of sand and vegetation areas (Table 1) and vegetation cover as the beach

Table 1
Summary of variables considered in this study.

Name	Description	Scale	Units
CI	Conservation index	–	1 to 10
RI	Recreation index	–	1 to 10
Sand	Sand area	On beach	m^2
Beach area	Sand + Vegetation area	On beach	m^2
Vegetation cover	Propotion of Beach area with a NDVI >0.25	On beach	o to 1
SST	Sea Surface Temperature (mean estimate for 2014–2018)	Nearest pixel	°C
SSTsd	Standard deviation in SST for 2014–2018	Nearest pixel	°C
HPD	Human Population Density	Vicinity	ind/m^2
NDVI	Normalized Differences Vegetation Index	Vicinity	−1 to 1
ALAN	Artificial Lights At Night	Vicinity	1 to 83
Seasonal tourism magnitude	Maximum Seasonality coefficient	Vicinity	–
Tourism effect	Total sum of Seasonality coefficient	Vicinity	–

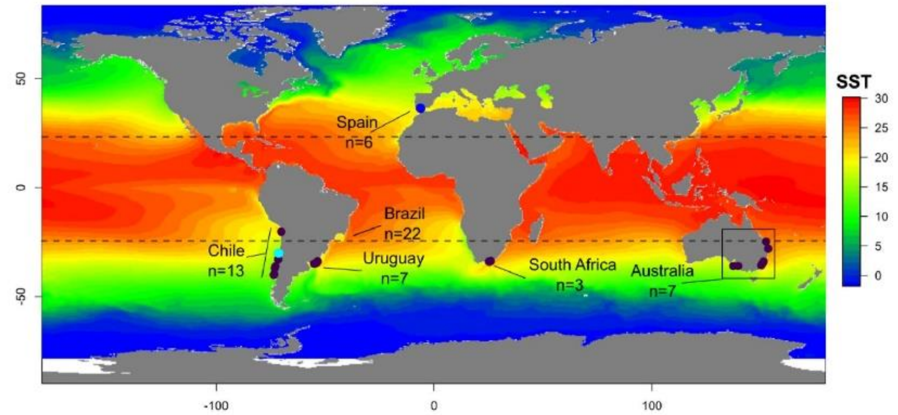

Fig. 1. Location of the 58 beaches from four continents included in this analysis, and the mean Sea Surface Temperature (SST) for the 2014–2018 period. Dot colors indicate different source publications (see Supplementary materials B and E for details on the database). (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

area's proportion covered by vegetation.

2.3. Sea surface temperature

Due to its ecological relevance and relationship with recreation, the mean and standard deviation of Sea Surface Temperature (SST) for the period 2014–2018 was incorporated into the analysis as a large-scale descriptor to allow comparisons between different geographic areas. This period was chosen to match the seasonality time frame (see below). Information was extracted from the pixel nearest to each site, using the NOAA Optimum Interpolation 1/4 Degree Daily SST Analysis Version 2 (Reynolds et al., 2007) (Table 1).

2.4. Urbanization and human use

Urbanization was measured on the beach vicinity using three indicators: human population density (McKinney, 2002), NDVI (Alberti, 2008), and ALAN (Zhou et al., 2014) (Table 1). Human Population Density (HPD) was estimated through the Gridded Population of World version 4 (CIESIN, 2017) and used as a direct measure of urbanization. Urban vegetation cover was estimated from Landsat 7 composites through NDVI. A mean annual NDVI was considered an inverse indicator of urbanization based on the assumption that densely urbanized areas have lower vegetation cover (Alberti, 2008). Annual mean ALAN estimates were extracted from the stable nighttime lights band of the Defense Meteorological Satellite (Baugh et al., 2010), with a spatial resolution of 1 km^2, and used as a proxy of urbanization due to its relationship with economic activity and urban infrastructure (Small and Nicholls, 2003).

Cumulative monthly ALAN values in a given area (Elvidge et al., 2014) were used to estimate monthly population variations by dividing each monthly ALAN value by ALAN on the month of the population census (Seasonality coefficient month $_{(i)}$ = Sum of ALAN month $_{(i)}$/Sum of ALAN month $_{(census)}$) (Stathakis and Baltas, 2018, see Supplementary materials C). This 'seasonality coefficient' was calculated in the beach vicinity using averaged monthly values for the period with available information (2014–2018) to provide a reliable indicator of monthly use of the urban infrastructure adjacent to beaches. As the seasonality coefficient is proportional to ALAN on the month of the population census, representing the baseline human disturbance, a value of 1 was subtracted from each month seasonality coefficient to calculate pulse estimators (Harris et al., 2018). The adjusted values for the seasonality coefficients were treated as a disturbance that influences sandy beaches conservation and recreation potential. Two pulse indicators were calculated to depict tourism seasonality: (1) seasonal tourism magnitude, as the maximum value of the seasonality coefficient; and (2) tourism effect, as the area under the seasonality coefficient curve (Jentsch and White, 2019).

2.5. Data analysis

A constrained envelope analysis (Marquet et al., 1995) was performed to determine the relationships of tourism indicators (seasonal tourism magnitude and effect) with SST and ALAN. The upper limit of the envelope corresponds to the maximum optimal combinations of the variables, with values above the upper ceiling showing an unusual relationship between them and values below the ceiling representing a wide range of suboptimal relationships (Caddy and Defeo, 2003). Nonlinear models were fitted for maximums of SST (categories of 2 °C) and ALAN (categories of 5 units), and the best ones were selected according to the coefficient of determination.

Considering all variables (Table 1), regression trees were built to identify meaningful beach attributes and determine how these variables relate to expert opinion on conservation and recreation potential (De'ath and Fabricius, 2000). Trees were constructed by iteratively partitioning the dataset into a nested series of mutually exclusive groups. Two Random Forests procedures (10,000 trees each) were trained, one for CI and another for RI, to evaluate each variable's importance and support the regression tree results. Three predictor variables were randomly chosen at each node and tried as splitting criteria (Liaw and Wiener, 2002). The variables predicted to be important in the model help understand which are associated with expert recommendations. The increase in mean squared error was reported related to prediction accuracy (Prasad et al., 2006).

Information on all spatial indicators included in the modeling process was obtained from global open databases through the Google Earth Engine (GEE) geographic information platform (Gorelick et al., 2017). Data analysis and visualization were performed through R software (R Core Team, 2020) and *rpart* (Therneau and Atkinson, 2019) and *randomForest* packages (Liaw and Wiener, 2002).

3. Results

Seasonality indicators exhibited similar patterns along with biogeographic and urban variables (Fig. 2). Only three sites (Pta. del Diablo, Uruguay; Totoralillo, Chile; and Coorong, Australia) were over the 0.6 maximum season magnitude, representing a 60% ALAN increase regarding the census month. These mid-latitude sites shared common

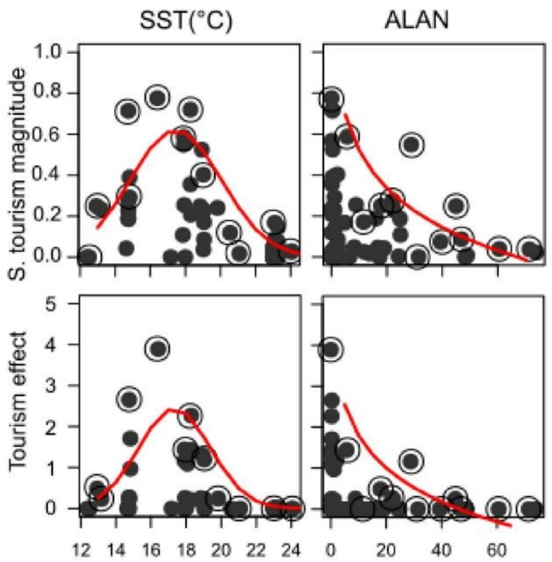

Fig. 2. Scatterplot showing the behavior of seasonal tourism magnitude (S. tourism magnitude) and tourism effect regarding mean Sea Surface Temperature (SST) and Artificial Lights at Night (ALAN). Significant results of the constrained envelope pattern analysis are shown: beaches used for model fitting are enclosed in circles, and red lines depict the fitted curves. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

features associated with high seasonality. The envelope analysis showed a bell-shaped pattern of seasonal tourism magnitude with SST ($r^2 = 0.71$, $p < 0.05$) and a decreasing trend with ALAN in the beach vicinity ($r^2 = 0.68$, $p < 0.001$). Tourism effect also showed a bell-shaped pattern with SST ($r^2 = 0.72$, $p < 0.05$) and an exponential decreasing trend with ALAN ($r^2 = 0.65$, $p < 0.001$) (Supplementary Material D).

The regression tree based on CI explained 73% of deviance and had a structure coherent with the index considerations on dune state, iconic species, and macrobenthic richness (Fig. 3a). ALAN was the dominant variable, and a higher conservation value was estimated for beaches with ALAN values < 3.6. This branch was further subdivided by vegetation cover: beaches with vegetation cover >18% had the highest CI values (average of 7.7). The Random Forest analysis explained 62% of the variance (Fig. 3b). ALAN was the most significantly associated with expert advised conservation priorities, followed by HPD and beach and sand areas.

The RI regression tree explained 49% of deviance and showed an unbalanced structure (Fig. 4a). The first split depended on seasonal tourism magnitude, with beaches with low seasonality (close to zero) displaying less RI values. Sites with ALAN >5.2 were associated with higher RI values. This branch was further subdivided, and beaches with ALAN >18 had the highest RI values. Beaches with some seasonality and ALAN <5.2 were further subdivided by vegetation cover, where less vegetated beaches showed higher RI scores. The Random Forest analysis explained 23% of the variance (Fig. 4b). RI was mainly explained by seasonal tourism magnitude and ALAN in the beach vicinity; vegetation cover, sand area, and HPD were also relevant.

4. Discussion

This paper reinforces the notion that sandy beach management needs to incorporate measurable variables that objectively describe human behavior (Fanini et al., 2020; Orlando et al., 2020). ALAN and seasonal tourism magnitude were the main variables explaining expert opinion on conservation and recreation indices. Both anthropogenic variables confirm human activity as a dominant driver behind management recommendations. The high explained deviance obtained by modeling expert opinion suggests a high potential for developing sandy beach management frameworks that incorporate openly available quantitative indicators.

Seasonality indicators showed coherence with local conditions (SST) as well as urbanization patterns (ALAN). The envelope analysis showed

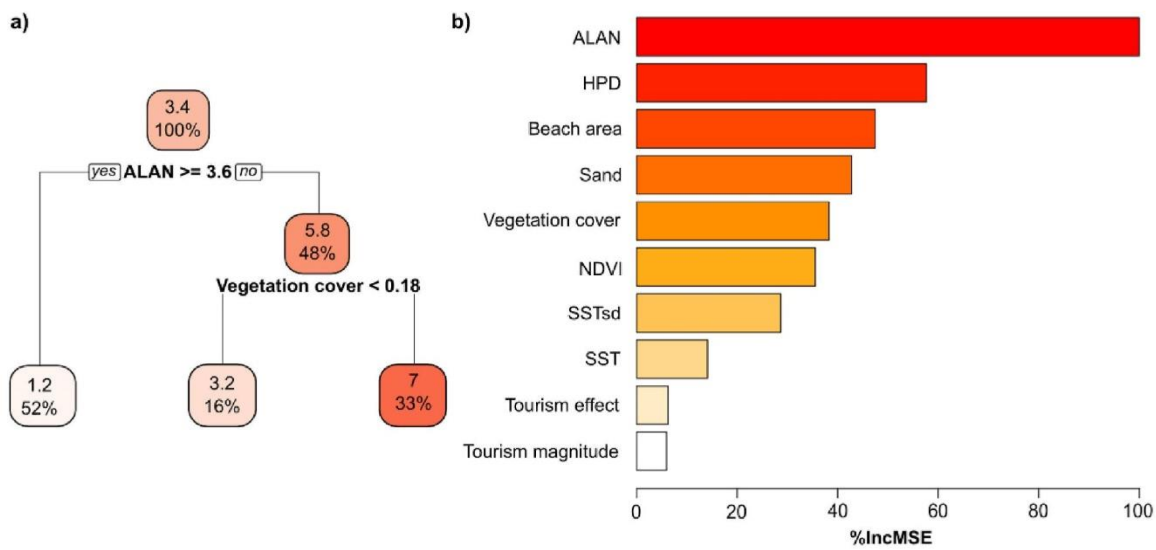

Fig. 3. Graphical representations of classification procedures of the Conservation Index. a) Regression tree showing the hierarchical relationship between the most relevant variables in beach classification. Splitting criteria are shown below each node; box numbers are the average value for the Conservation Index and the percentage of data at each node. b) The importance of quantitative indicators determined by Random Forests as percentual increases in mean square error (%IncMSE) at each split: each variable estimate is given by the difference between the residual sum of squares before and after the split. 'Tourism magnitude' refers to seasonal tourism magnitude, ALAN to Artificial Lights At Night, HPD to Human Population Density, SST to Sea Surface Temperature, and NDVI to the Normalized Differences Vegetation Index.

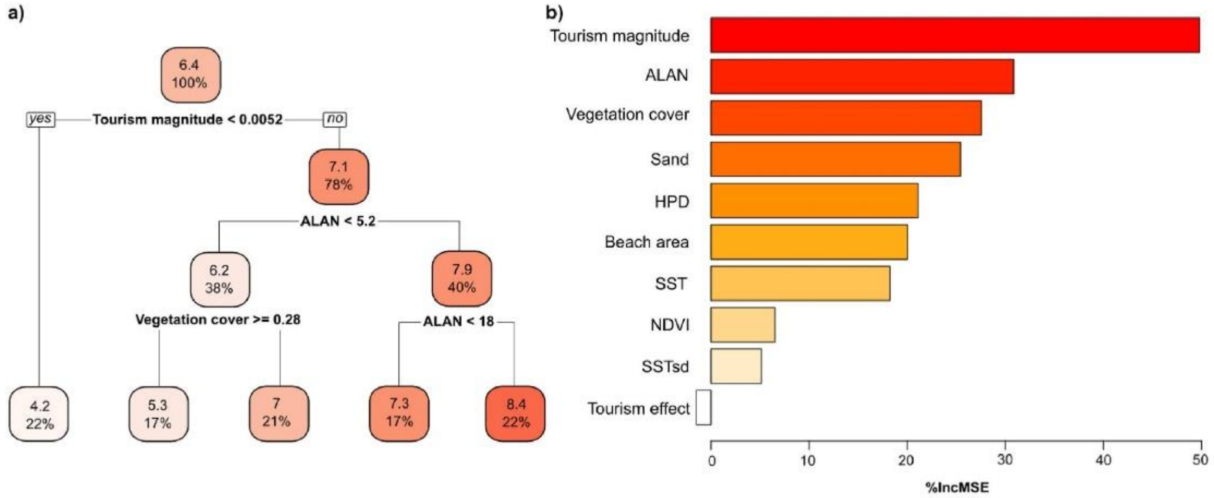

Fig. 4. Graphical representations of classification procedures of the Recreation Index. a) Regression tree showing the hierarchical relationship between the most relevant variables in beach classification. Splitting criteria are shown below each node; box numbers are the average value for the Recreation Index and the percentage of data at each node. b) The importance of quantitative indicators determined by Random Forests as percentual increases in mean square error (%IncMSE) at each split: the value shown for each variable is the difference between the residual sum of squares before and after the split. 'Tourism magnitude' refers to seasonal tourism magnitude, ALAN to Artificial Lights At Night, HPD to Human Population Density, SST to Sea Surface Temperature, and NDVI to the Normalized Differences Vegetation Index.

that SST constrain tourism seasonality with a maximum close to 17 °C. High SST generates low seasonality due to year-round acceptable recreational conditions, while low SST inhibits beach tourism. Intermediate values show higher seasonality that characterizes temperate latitude beaches, reinforcing the importance of the seasonality indicators proposed here. Urbanization density sets the human carrying capacity (accommodation) at a given area and can be estimated through ALAN. However, seasonal use adds complexity to the ALAN-urbanization relationship, as beaches with urbanized vicinities that receive seasonal tourism should present high seasonality and low ALAN. By contrast, beaches with dense and stable populations in their vicinity should present low seasonality and high ALAN, due to high all-year-round light emission.

The constrained envelope analysis showed an exponential decrease along the ALAN axis for both seasonality indicators (Fig. 4), describing a dynamic in which resident and seasonal use shape ALAN emissions. Sites under the upper ceiling have an expected relationship between stable and seasonal ALAN. In contrast, those above the envelope have a very high level of seasonal use, indicating highly touristic sites or alternative forms of seasonal use. Describing the relationship between seasonal and stable use gives further insight into the interaction between urbanization and resource use. This is particularly relevant, as the interactions among the triple whammy threats (urban and industrial development, use of resources, and the effects of climate change) and the mechanisms operating behind them are still poorly understood (Defeo and Elliott, 2021).

Mixed-use management of sandy beaches implies simultaneous resource exploitation, recreation, and conservation issues (McLachlan et al., 2013). This approach is generally adopted when conservation and recreation values are similar. However, this middle ground option has been hard to define and implement. The seasonality index (Stathakis and Baltas, 2018) and the tourism indicators presented here allow to delineate management plans regarding human use variations throughout the year. A seasonal mixed-use management approach can be defined by adding a temporal dimension to the open concept of mixed use, delimiting seasonal priorities to maximize conservation and recreation results. Further development of this management concept is relevant, particularly for mid-latitude beaches, as the context of global coastal urbanization is likely to lead to a generalized scenario of mixed use sandy beach.

The regression tree showed that satellite-based indicators have a good explanatory power of expert opinion on conservation value. The high relevance of ALAN and vegetation cover reinforces the importance of human use patterns for management and is following CI considerations: 1) pristine and extensive dunes are associated with low urbanization (ALAN) and high beach vegetation (Defeo et al., 2009); 2) endangered and iconic species relate with ALAN by stress and habitat reduction (Duarte et al., 2019; Gaston et al., 2013; Quintanilla-Ahumada et al., 2021), while higher vegetation cover implies the availability of more habitats; 3) there is a negative relationship between ALAN and macrobenthic diversity on sandy beaches (Orlando et al., 2020).

The Random Forest results show higher importance of urbanization (ALAN, HPD) over ecosystem characteristics (beach area, sand, and vegetation cover) regarding conservation. This general pattern coincides with the regression tree structure, further reinforcing the importance of human activity as the primary driver of conservation recommendations. The vegetation cover indicator does not inform the composition of the vegetation; therefore, high values could be associated with invasive vegetation. A finer scale analysis of beach vegetation composition is possible using Landsat information (César de Sá et al., 2017), but this kind of approximation should be site-specific. For large-scale comparisons and rapid assessments, vegetation cover can be complemented with the dune state indicator included in the CI (McLachlan et al., 2013).

Although tourism seasonality was not explicitly considered in the RI developed by McLachlan et al. (2013), the regression tree and the Random Forest analyses identified tourism magnitude as the most informative variable in predicting expert opinion on recreation potential, supporting the relevance of developing seasonal mixed-use based on quantitative indicators. Recreation potential was also associated with low vegetation cover and high ALAN. These features are generally associated with urbanized beaches, confirming the relevance of recreation on increasing coastal urbanization (Defeo and Elliott, 2021). The lower percentage of variance explained by both RI approaches can be explained by the lack of a direct estimation for two of its components: the extent of recreation infrastructure and the level of safety/health of a beach. The 'physical carrying capacity' component in the RI index can be related to sand area. This indicator showed high relevance in the Random Forest analysis and directly quantified a key management aspect. However, sand area is a complex indicator because high values could be related to pristine sites with good conservation potential or beaches with high recreation potential subjected to nourishment events.

The rapid urban population growth and coastal infrastructure development worldwide add dimensionality and complexity to sandy beach management (Schlacher et al., 2007; McLachlan and Defeo, 2018). Addressing this challenge requires integrated approaches that consider several drivers, pressures, and interests from different disciplinary angles (Groeneveld et al., 2018). The quantification and inclusion of tourism seasonality constitute the first step toward developing more rigorous approaches to assess the complexity behind the dynamics of sandy beach use and its environmental consequences (Fanini et al., 2020). The results reinforce the idea that sandy beach management should incorporate quantitative ecosystem attributes, including the long-postponed matter of tourism seasonality. Satellite-based information used here can increase the rigorousness and quality of management by better assessing human use patterns, incorporating quantitative and objective indicators that reduce the subjectivity that characterizes expert opinion.

In summary, the combined analysis of ALAN and seasonal tourism magnitude allowed a better understanding of the profile of use of each beach, providing a much-needed global and standardized context to assess urban impact. Satellite-based information may help compare different contexts at large geographic scales or monitor the progress of management and restoration initiatives beyond the inherent subjectivity of expert opinion. On the other hand, a strictly inductive method would lose any vision and innovation potential, which require expert knowledge at different spatial and temporal scales. Therefore, this combined and complementary approach merging scientific information derived from satellite-based information (this paper) and expert knowledge (McLachlan et al., 2013) could lead to a coherent management narrative about the potential use of beaches for recreation and or conservation. This innovative perspective based on a mixed-methods approach could also be useful to include different biophysical and human attributes to provide an incremental spectrum of management actions needed to sustain beach ecosystem services over time. Integrating different formats of knowledge could increase the ability to explore, promote, strengthen and support management strategies to improve the status of a fragile ecosystem at risk increasingly threatened by several stressors acting together.

CRediT authorship contribution statement

L. Orlando: Conceptualization, Methodology, Software, Visualization, Writing – original draft. **L. Ortega:** Conceptualization, Formal analysis, Supervision, Writing – review & editing. **O. Defeo:** Conceptualization, Investigation, Supervision, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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7. DISCUSIÓN GENERAL

En este Capítulo se discuten los principales resultados del proceso de análisis sobre la relación entre la urbanización y la ecología de sistemas de playas. La discusión en el contexto de una tesis en ecología permite destacar resultados e hipótesis que trascienden a la publicación de trabajos científicos, pero hacen a la construcción de un marco para la investigación ecológica y el manejo en playas urbanas.

La correlación negativa entre el grado de urbanización y la diversidad de macrofauna describe la naturaleza de la relación entre urbanización y los ecosistemas de playas. El análisis de patrones de uso y variables satelitales complementa esta relación, abordando la diversidad de factores humanos y la importancia de los distintos modos de uso que reciben las playas. El análisis de forzantes climáticos sobre el área de playa aborda el componente físico del ecosistema y los impactos del cambio y la variabilidad climática. En suma, y de manera convergente, se realizaron aportes al análisis de los componentes del '*triple whammy*', incluyendo los efectos de la urbanización, el cambio climático y el uso de los recursos en los ecosistemas de playas.

7.1 Cambio climático

Las tendencias de largo plazo en el área de playa en la costa de Montevideo mostraron un ciclo de 27 años, con fases bien definidas de acreción y erosión. El nivel de resolución temporal (anual) junto con lo extenso de la base de datos (33 años), permitió establecer correlaciones con variables climáticas locales y fenómenos globales que afectan la configuración climática en la región. Investigaciones previas basadas en fotografías aéreas mostraron una tendencia erosiva de largo plazo para algunas playas de esta costa (Gutiérrez *et al.* 2006), esta información independiente fue utilizada para estimar la precisión de las medidas obtenidas mediante el proceso semi automatizado y realizar la validación del método (Anexo I).

El modelo desarrollado confirmó la relevancia de la dirección del viento predominante como forzante del área de las playas, lo cual coincide con la importancia del viento como variable dominante en el Río de la Plata (Barreiro *et al.* 2017). Además, este modelo puede alimentarse con información de variables climáticas estimadas en distintos escenarios de cambio climático para predecir la evolución de la costa de Montevideo. Esto permitiría detectar las zonas con mayor grado de erosión proyectada y tomar decisiones de manejo con una perspectiva temporal adecuada, atendiendo a la dinámica de estos sistemas. El Capítulo 4 aborda el estudio de la dinámica a nivel de costa, y por tanto el desarrollo de modelaos para playas individuales puede proveer soluciones a medida en sitios objetivo. La metodología aplicada y la base de datos actualizada hasta 2019 (Anexo I) constituyen un insumo valioso a la hora de desarrollar planes de manejo o realizar estudios de dinámica costera.

Los cambios asociados a la urbanización afectan las tasas de erosión de las playas por alteraciones producidas sobre tierra y en el agua. En tierra se producen cambios en la cobertura del suelo, interrupción de los flujos de arena generados, canalización de cursos de agua e incremento de efluentes (Short 1999, Jackson & Nordstrom 2011). El desarrollo costero tiene asociado la construcción de estructuras duras que afectan la circulación del agua y la arena, así como muros de contención que limitan el desplazamiento del ecosistema de playa hacia tierra (Pontee 2013). Si bien los efectos concurrentes de la urbanización y el cambio climático no fueron analizados de manera específica, con la metodología desarrollada y los indicadores urbanos evaluados en esta tesis, es posible abordar esta problemática y comparar el comportamiento de ecosistemas con diferentes grados de urbanización y modos de uso. Esto constituye un insumo relevante para el estudio de la ecología en playas urbanas y su manejo.

Actualmente existe un debate sobre el futuro de las playas arenosas debido al aumento del nivel del mar. Estudios globales las señalan como ecosistemas en peligro de extinción por dicho aumento (Vousdoukas et al. 2020), aunque otros trabajos indican que muchas playas contemporáneas se formaron hace cientos de años y migraron hacia la tierra durante las épocas post glaciares y no han desaparecido (Cooper *et al.* 2020). De este argumento se desprende que la mayor amenaza para la existencia de las playas son las estructuras duras de defensa costera que limitan la dinámica de las playas (Cooper *et al.* 2020). Desde una perspectiva humana, los sistemas de mayor vulnerabilidad son aquellos de mayor importancia directa, porque generan desarrollo económico y por la presencia de viviendas e industrias que aumentan la importancia de los servicios ecosistémicos locales. Esta amenaza implica una mayor urgencia para la incorporación cuantificada de la urbanización en los estudios ecológicos y planes de manejo en ecosistemas de playas arenosas.

7.2 Urbanización y biodiversidad

A pesar de la extensión y el rápido desarrollo del análisis de patrones ecológicos asociados a la urbanización, existen muchas preguntas abiertas sobre el efecto de diferentes grados de urbanización sobre la riqueza de especies (Piano *et al.* 2019). En general se abordan comunidades insertas en la matriz urbana y se obtienen patrones de variación respecto a diferentes grados de cobertura urbana del suelo (Alberti 2008). Tanto la movilidad del taxón utilizado, como la comparación a través de hábitats con cualidades diferentes (bosques, cultivos, urbanización), generan una gran variación entre muestras que puede enmascarar el efecto de la urbanización (McDonnel & Hahs 2008). Por ejemplo, la riqueza de aves y mariposas muestra un máximo a niveles intermedios de urbanización (Hansen *et al.* 2005), mientras que la de anfibios responde más fuertemente a la conectividad entre parches y las extinciones locales que al grado de urbanización (Pillsbury *et al.* 2008). Además, la urbanización altera los patrones de diversidad, el cambio en las condiciones ambientales puede generar el aumento en la densidad de especies que se favorecen de las condiciones urbanas asi como la introducción de especies exóticas (Gaston *et al.* 2010). Esta variedad de patrones resalta la necesidad de abordar los distintos mecanismos que afectan las respuestas de las especies a la urbanización (Piano *et al.* 2019). El diseño del análisis realizado en el Capítulo 5 sobre un mismo tipo de ecosistema, y en una comunidad donde no se ha registrado sustitución de especies, minimiza las variaciones entre muestras y permite comparar la relevancia de las variables urbanas, contrastándolas con las variables físicas que clásicamente han explicado la riqueza macrofaunística en playas arenosas.

Las zonas de barrido de ola más amplias se asocian a climas de swash más benignos, en los que ocurren alta diversidad y abundancia de macrofauna. Este patrón está tan bien documentado a nivel mundial que se ha transformado en un paradigma (Jaramillo & McLachlan 1993). La severidad del clima de swash junto con las interacciones bióticas constituyen el marco predictivo general para la macrofauna en playas arenosas, utilizándose para estimaciones de abundancia y diversidad (Defeo *et al.* 2003, Lercari & Defeo 2006, Barboza & Defeo 2015). El análisis de la riqueza macrobentónica (Capítulo 5) indica que la densidad de población humana supera en capacidad explicativa al ancho de la zona de swash. Este resultado, si bien es local, tiene implicaciones profundas para la ecología de playas en particular y la ecología en general. La urbanización afecta tanto la severidad del ambiente como las interacciones bióticas (Alberti 2008), y al no incluir específicamente al ser humano se omite la fuente de disturbio más relevante en el Antropoceno (Worm & Paine 2018). Los resultados evidencian que los paradigmas ecológicos en playas arenosas mantienen su validez, pero deben ser actualizados. El desarrollo de medidas cuantitativas, disponibles a nivel global, para estimar la urbanización constituye un paso en esta dirección y es relevante para la ecología en el contexto de un planeta dominado por humanos (Alberti 2008, McDonnel & Hahs 2008, Pillsbury *et al.* 2008, Defeo *et al.* 2009, Worm & Paine 2018).

En este trabajo se emplearon indicadores que describen la urbanización bajo la premisa de que ésta es una variable compleja que resume cambios en la cobertura del suelo, aumento de efluentes, prevalencia de especies exóticas y elevada presencia humana, entre otros disturbios, discutidos en el Capítulo 5 (Alberti 2008, Zhou *et al.* 2014). Esto implica suponer patrones regulares de impacto a través de diferentes culturas e infraestructuras, algo que ha demostrado no ser enteramente válido (Sazatornil *et al.* 2016). El impacto humano es moderado por componentes culturales y físicos, por lo cual resolver o minimizar los conflictos entre las personas y los ecosistemas es urgente en el contexto actual.

7.3 Manejo de ecosistemas de playas

Millones de visitantes eligen visitar estacionalmente las playas para disfrutar sus servicios de recreación (Fanini *et al.* 2020). De este patrón, sumado a la preferencia humana por establecerse en la costa (Small & Nicholls 2003), se desprende la necesidad de repensar la relación humana con las playas arenosas en un contexto global y de manera acorde a la relevancia de los impactos de las acciones humanas. Esto refuerza la noción de que el manejo de los ecosistemas de playas debe incorporar variables cuantitativas, así como indicadores que describan objetivamente los patrones de uso humano (Fanini *et al.* 2020).

El metaanálisis de recomendaciones de uso y su contraste con variables urbanas y ecológicas (Capítulo 6) permitió contextualizar el componente humano, atendiendo a formalizar el estudio de su impacto sobre estos ecosistemas. Las luces artificiales nocturnas (ALAN) y la magnitud del turismo estacional fueron las principales variables explicativas de los valores de conservación y recreación percibidos por expertos. Ambas variables son antropogénicas, señalando a la actividad humana como un forzante relevante a la hora de generar recomendaciones de manejo. Este resultado indica que el manejo y la clasificación de playas de acuerdo con su uso prioritario están condicionados por el componente humano. Lo anterior hace surgir la pregunta ¿la conservación es priorizada donde se debe o donde se puede? Más allá de la cuestión filosófica de fondo, la incorporación de variables objetivas llevada adelante en esta tesis aporta elementos de importancia para detectar este efecto y sentar las bases para un manejo mejor fundamentado. Los resultados refuerzan la importancia de variables ecológicas clásicas como la cobertura vegetal para la conservación, y la capacidad de carga para la recreación. Los altos porcentajes de varianza explicada sugieren un buen potencial para el desarrollo de aproximaciones de manejo que incorporen indicadores cuantificados y de acceso abierto. Además, se generó un marco estandarizado de la relación entre urbanización y uso estacional que permite incorporar el turismo como variable ecológica y de manejo en estos ecosistemas.

Las ciudades constituyen un laboratorio natural para el estudio de las dinámicas eco-evolutivas (Alberti *et al.* 2017). El análisis de los patrones de uso de la urbanización, y en particular del modelo de turismo estacional, abre una serie de posibilidades en cuanto a estudios en ecología y evolución. El desarrollo urbano modifica las condiciones bióticas y abióticas, creando nuevas presiones de selección (Johnson & Munshi-South 2017). Repensar los procesos evolutivos en un planeta urbanizado requiere reconocer las interacciones reciprocas entre selección y cambios ambientales (Alberti *et al.* 2017). Existe evidencia de que las ciudades generan aumentos en tasas microevolutivas, por lo cual la urbanización puede afectar los ecosistemas causando cambios sistémicos en los rasgos funcionales que regulan la productividad y estabilidad de dichos ecosistemas (Alberti *et al.* 2017). Las alteraciones en la diversidad local mediadas por la urbanización pueden influir en procesos evolutivos y ecológicos de mayor escala, a través de cambios en la competencia y en las interacciones depredador-presa. La urbanización con residentes estables ejerce un disturbio de presión, generado por la alteración sostenida de los parámetros ambientales; este tipo de disturbio generalmente ocasiona degradación en los ecosistemas (Harris *et al.* 2018). Alternativamente, en costas con uso estacional, el impacto humano directo es mayor durante el verano, funcionando como un disturbio de pulso que puede desencadenar cambios en el estado ecosistémico (Jentsch & White 2019). Estos patrones de uso se reflejan en presiones eco evolutivas disímiles sobre ecosistemas similares, lo que puede generar patrones de alteración diferentes. Es importante identificar cambios en las dinámicas eco evolutivas a nivel comunitario en ambientes urbanos como aporte de casos de estudio, ya que esta temática ha sido estudiada principalmente a través de simulaciones (Uchida *et al.* 2020). En este sentido, las costas urbanizadas de uso estacional constituyen un experimento específico donde el impacto humano directo esta magnificado durante la temporada de verano. Esto puede producir patrones evolutivos divergentes entre playas de uso estable y playas de uso estacional, tanto a nivel específico como en la estructura de los ecosistemas.

7.4 Conclusiones generales y perspectivas

Dada la relevancia de las playas arenosas como proveedoras de servicios ecosistémicos y sus implicaciones socio-económicas, es imperativo el desarrollo del conocimiento sobre las dinámicas urbanas costeras y sus impactos ecológicos si se pretende mitigar los impactos humanos sobre las costas. Los resultados derivados de esta tesis permiten, en conjunto, un abordaje más riguroso de cualquier ecosistema de playa, destacándose:

- El análisis histórico del área de playa y la determinación de los forzantes climáticos más importantes, posibilitando la evaluación de tendencias históricas, así como análisis predictivos de los potenciales efectos del cambio climático.

- La cuantificación e incorporación del factor urbano a los estudios ecológicos y una evaluación de su impacto sobre la comunidad macrobentónica, clave en estos ecosistemas.

- Un contexto estandarizado para el abordaje del manejo a través de variables satelitales cuantificadas que abordan la importancia de los distintos tipos regímenes de uso.

Estos resultados permitieron mejorar las capacidades de análisis y aportar mayor claridad sobre algunas de las dinámicas más relevantes de la actualidad, cumpliéndose con el objetivo general. Como perspectivas de trabajo a futuro se destacan: 1) el análisis macroecológico a escala global de los patrones de riqueza de macrofauna incorporando la urbanización y los patrones de uso humano como variables explicativas; y 2) el desarrollo de bases de datos de área de playa que recuperen información de los últimos 40 años y actúen como cimientos para una mejora en el conocimiento y la gestión de la dinámica costera. Ambas perspectivas aportan a la construcción de un marco teórico y de manejo que integre plenamente la actividad antrópica en la ecología de playas, lo que resulta imprescindible para el siglo XXI.

En el caso de Uruguay, las playas han sido principalmente condicionadas para proveer infraestructura, calidad estética y servicios para el desarrollo de actividades turísticas y de descanso. Sin embargo, muchas áreas desarrolladas para turismo han experimentado cambios hacia un uso estable, aprovechando la infraestructura desarrollada para el turismo. Esto lleva a que coexistan diferentes niveles de perturbación dados por la combinación de efectos de pulso (turismo estacional) y presión (e.g. limpieza, contaminación lumínica, etc.), que afectan negativamente algunos de los elementos ecológicos y naturales de las playas. La mejor estrategia para la conservación de estos ecosistemas costeros residiría en comprender que la disminución de su oferta a nivel mundial, por efecto combinado del cambio climático e intervenciones humanas, seguramente se refleje en un aumento del valor turístico de playas prístinas. Esto implicaría generar pautas de manejo tendientes a asegurar la conservación y la sostenibilidad de los servicios asociados, que consideren los diferentes niveles de urbanización y los tipos de uso. Esto requiere una estrategia de desarrollo costero coordinada y políticas de Estado asociadas a diferentes niveles (anidados) de toma de decisiones (e.g. departamental, nacional).

La falta de interacción entre los distintos niveles de toma de decisiones en el manejo de recursos naturales, y la ausencia de aplicación del conocimiento producido por la investigación científica, son problemas que debieron ser abordados hace años. Sin embargo, aún debe insistirse en la incorporación de investigación científica a los procesos de gestión de los ecosistemas de playas, tan relevante para nuestro país y el mundo. Es necesaria una mayor participación de las instituciones que realizan investigación en el manejo de los ecosistemas de playas para mitigar los efectos de los eventos climáticos extremos, la urbanización y los distintos tipos de uso. Para esto es indispensable invertir en investigación para afrontar los futuros desafíos que convergen en una mayor presión humana y la degradación del ambiente, en un contexto climático adverso.

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9. ANEXO I: ESTIMACIÓN DEL ÁREA DE PLAYA MEDIANTE INFORMACIÓN SATELITAL DE ACCESO ABIERTO: UNA CALIBRACIÓN PARA LA COSTA DE MONTEVIDEO, URUGUAY

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Sandy beach area estimation through open access satellite information: A calibration for the coast of Montevideo, Uruguay

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ABSTRACT

Sandy beaches provide a wide variety of ecosystem services that support human well-being at coastal areas. These ecosystems are highly dynamic and primarily defined by the interaction between wave energy, tides and wind regimes. High variability makes beaches vulnerable to physical modifications and climate change, jeopardizing ecosystems functions and sediment cycles. Resulting in accelerated erosion rates and ecological degradation with widespread socioeconomic implications. Coastal analysis is a data demanding process that requires long term monitoring programs; this study applies an open access methodology, to yearly composites of satellite images from the Landsat collection, in order to estimate beach area. This informative variable can help elucidate coastal dynamics, ecosystem attributes and touristic potential. Sand and vegetation cover were considered as components of the beach ecosystem, sand was detected through a random forest semi-automated classification, and vegetation area was estimated by applying a threshold to the normalized difference vegetation index. The method was validated, and a calibration of the parameters was performed for the Montevideo coast, by testing results against independent estimations of beach area based on aerial and satellite imagery. Using the best performing parameters the area of 20 beaches of the coast of Montevideo was estimated for a 35 years period. This methodology can be applied anywhere at a very low operational cost, potentially multiplying the available information and knowledge on the pressing matters of coastal dynamics and sandy beach management.

KEY WORDS: Sandy beach, Satellite Information, Open access, Ecosystem management,

Coastal erosion

RESUMEN

Las playas arenosas brindan una amplia variedad de servicios ecosistémicos que sostienen el bienestar humano en áreas costeras. Estos ecosistemas son muy dinámicos y dependen fuertemente de la interacción entre la energía de las olas, las mareas y los regímenes de viento. Su alta variabilidad hace que las playas sean vulnerables a las modificaciones físicas y el cambio climático, poniendo en peligro la estabilidad de las funciones ecosistémicas y los ciclos de los sedimentos. Esta fragilidad ha resultado en tasas de erosión y degradación socioeconómica y ecológica aceleradas. El análisis de la dinámica costera demanda una gran cantidad de información, este estudio aplica una metodología de acceso libre para estimar el área de playa a través de mosaicos anuales de imágenes satelitales de la colección Landsat. El área de las playas es una variable relevante para dilucidar la dinámica costera, los atributos ecosistémicos y el potencial turístico. La arena y la cobertura vegetal se consideraron como componentes del ecosistema de playa, la arena se detectó mediante una clasificación semi automatizada de tipo Random Forest, mientras que el área de vegetación se estimó aplicando un umbral al índice de diferencias normalizadas de vegetación (NDVI). Se realizó una validación del método, y se calibraron los parámetros de estimación para la costa de Montevideo, comparando los resultados con estimaciones independientes de área de playa basadas en imágenes aéreas y satelitales. Utilizando los parámetros de mejor desempeño, se estimó el área de 20 playas de la costa de Montevideo para un período de 35 años. Esta metodología abierta está disponible para su aplicación en cualquier ubicación a un costo operativo muy bajo, lo que implica una oportunidad potencial para multiplicar la información disponible y el conocimiento sobre dinámica costera y manejo de playas arenosas.

PALABRAS CLAVE: Playas arenosas, Información satelital, Acceso libre, Manejo ecosistémico, Erosión costera

INTRODUCTION

Sandy beaches comprise more than two thirds of the ice-free coasts of the world, providing a wide variety of ecosystem services that support human well-being at coastal areas (McLachlan and Defeo, 2018). Beaches are highly dynamic and primarily defined by the interaction between wave energy, tides and wind regimes (Barnard *et al.*, 2015; McLachlan *et al.*, 2018), which translates into strong variation patterns of ecosystem area (Short, 1999). High variability makes beach ecosystems particularly vulnerable to physical modifications, armoring structures and other urban-associated alterations, which jeopardize ecosystems area, reduce biodiversity and alter sediment budget cycles, accelerating erosion rates (Short, 1999; Defeo *et al.,* 2009). Coastal ecosystems worldwide face a “triple whammy” given by increases in urban and industrial development, use of resources, and the effects of climate change (Defeo and Elliot, 2020). Natural and anthropogenic stressors imposed on sandy beaches have led to widespread erosion and degradation of these complex ecosystems with consequences for both social and ecological components (Amyot and Grant, 2014; Luijendijk *et al.*, 2018).

The mitigation of beach ecosystem degradation is a complex task that requires morphological estimations to assess drivers and trends (Barnard *et al.*, 2012). On-site coastal monitoring programs are expensive and often sparse (Splinter *et al.*, 2013; Vos *et al.*, 2019). This is especially problematic, as robust long-term data sets are required before meaningful trends emerge (Short and Jackson, 2013). Multi-decadal analysis of coastal vulnerability have shown oscillation patterns and correlations to global and regional climatic indices, in combination with predominant wave climate and sea level (Barnard *et al.,* 2015; Orlando *et al.,* 2019). This climate-driven dynamic supports the shared concern about the potentially critical effects of climate change at coastal areas, reinforcing the need for long-term coastal databases.

Satellite imagery provides useful data for studies of near shore morphodynamics (Harris *et al.*, 2011; Short and Jackson, 2013) and has been used for shoreline and coastal monitoring at global (Luijendijk *et al.*, 2018, Vos *et al.*, 2019) and local (Cifuentes *et al.,* 2017) scales. The Landsat satellite collection has spatial, spectral, and radiometric resolution that, along with their temporal continuity, have proven well suited for beach erosion monitoring (Luijendijk *et al.*, 2018, Orlando *et al.,* 2019; Vos *et al.*, 2019).

This study proposes an open access methodology to estimate beach area with the objective of strengthening coastal management capacities by providing a low cost remote sampling procedure. The approach focuses on beach area determination, an informative variable that can help elucidate coastal dynamics (Short and Jackson, 2013; Orlando *et al.*, 2019), ecosystem attributes (Takimoto and Post, 2013) and touristic potential (Mclachlan *et al.*, 2013). Sand and vegetation cover are considered, sand detection is achieved through a Random Forest semi-automated classification, while vegetation area is estimated by applying a threshold to the normalized difference vegetation index (NDVI). As a case study, calibration of the estimation parameters is performed for the Montevideo coast by testing results against previous estimations of beach area based on aerial and satellite imagery (Gutiérrez *et al.*, 2016). Then, using the best performing parameters, beach area is estimated for 20 beaches of the coast of Montevideo for the 1984-2019 period.

STUDY AREA

Uruguay is located in the southeastern coast of South America, with a total area of approximately 176,000 km^2 (Fig. 1). Montevideo is the administrative department that holds the capital city and main port, its coast is characterized by sandy beaches interrupted by rocky heads, with a semidiurnal tidal regime of microtidal amplitude (ca. 0.5 m) (Lercari and Defeo, 2006). The Rio de la Plata system has a major freshwater input from Parana and Uruguay rivers, from the west, forming a shallow (up to 15 m) coastal-plain estuary (Lercari and Defeo, 2006). A strong turbidity front is located around Montevideo city, which constitutes the surface indication of the transition between fresh and saline waters (Sepúlveda *et al.*, 2004). In this estuary, variability of water characteristics (salinity, temperature and turbidity) is mainly forced by winds (Simionato *et al.*, 2010), strong onshore winds also produce short-term increases in sea level. The frequency of storm surges greater than 200, 250 and 280 cm above mean sea level has increased and is attributed to changes on wind regimes (Verocai *et al.* 2015).

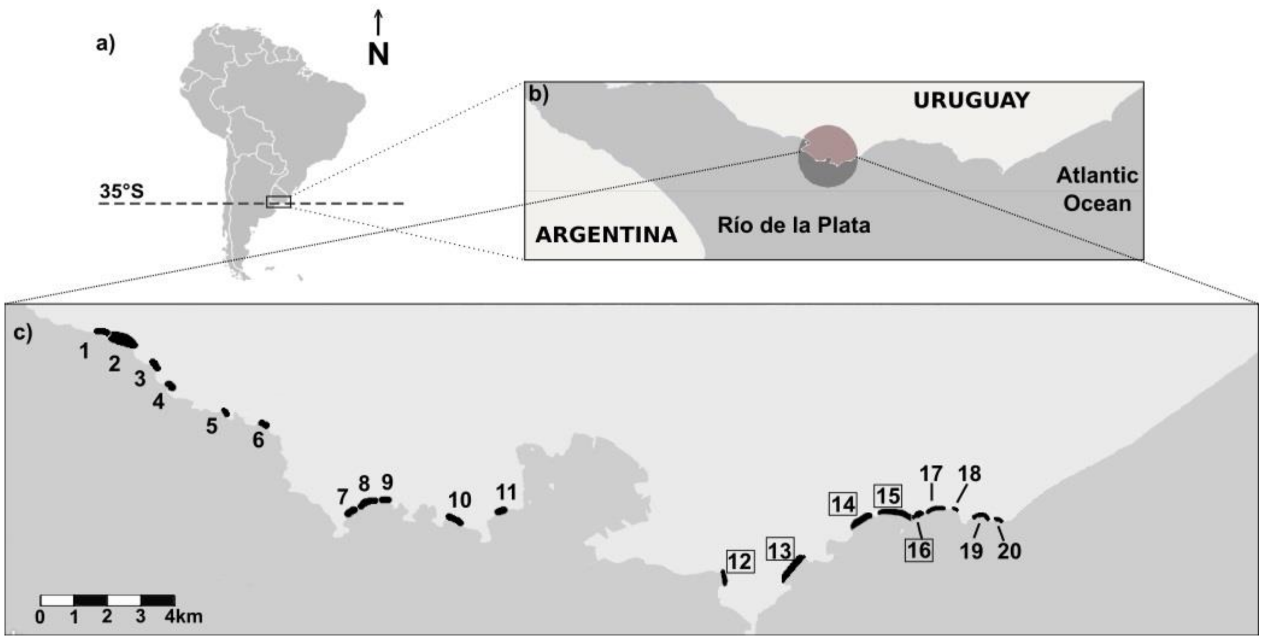

Figure 1. a) Location of study site on South America. b) Rio de la Plata, the circle indicates the location of Montevideo. c) The 20 beaches selected as study sites. From left (west) to right (east): 1) Punta Espinillos, 2) Mailhos, 3) Rocha, 4) La Colorada, 5) Juan Torora, 6) Pajas Blancas, 7) Punta Yeguas chica, 8) Punta Yeguas grande, 9) Santa Catalina, 10) Del Nacional, 11) Cerro, 12) Ramírez, 13) Pocitos, 14) Buceo, 15) Malvín, 16) Brava, 17) Honda, 18) Ingleses, 19) Verde, 20) La Mulata. Squares enclosing beach numbers indicate locations that were used for the calibration procedure.

MATERIALS AND METHODS

Beach area was estimated yearly from 1984 to 2019 for 20 beaches of the Montevideo coast (Fig.1c), beaches are referred by common name and were selected, not as an exhaustive list but, to cover the geographical range of this coast. The Landsat 5 and 7 satellite imagery collections were analyzed through Google Earth Engine (GEE) open platform (Gorelick *et al.*, 2017). This cloud-based platform for geospatial analysis uses a WGS84 projection and allows to access and analyze georeferenced satellite collections remotely using Google´s computation infrastructure, thus reducing the hardware requirements for the analysis and avoiding the storage of images.

The Landsat satellite series has a temporal granularity of 16 days, Landsat 5 has been active from 1984 to 2012, comprises 7 spectral bands with a maximum spatial resolution of 30m, covering different wavelengths from the blue range (0.45 - 0.52 µm) to the shortwave infrared (2.08 - 2.35 µm) (Ozturk and Sesli, 2015). The Landsat 7, comprises 8 spectral bands ranging to blue (0.45 -

0.52 µm) to shortwave infrared (2.08 - 2.35 µm) with a panchromatic band (0.52 - 0.90 µm) and a maximum spatial resolution of 15m (Gorelick et al., 2017).

The applied methodology combined photo interpretation with machine learning algorithms to classify land cover (Fig. 2). The detailed explanation of the methodology applied is referred to Figure 2. Each numbered paragraph refers to a box representing a stage of the process. When applies, below the numbered paragraphs there is information on the particular case for the Montevideo coast.

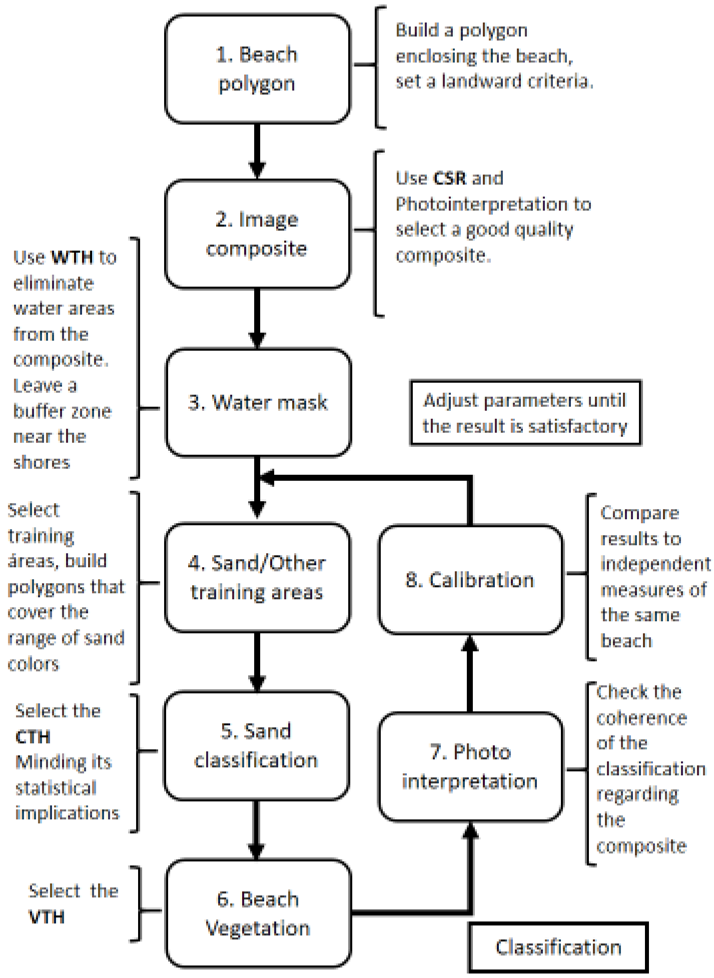

Figure 2. Work flow of the methodological approach applied to measure the area of sandy beaches. Each box represents a step with referenced additional information. CSR is the cloud score range, WTH is the water threshold, CTH is the classification threshold and VTH the vegetation threshold.

1. Beach polygon. A polygon enclosing each of the beaches was constructed based on Quickbird satellite images from Google Earth (dated July 9 of 2009, 2.5m spatial resolution), and used to obtain measures for each beach. The seaward limit of the polygon should be able to capture area variations and different beach profiles without having to be modified (Fig. 3). The landwards limit in urban beaches is set by hard structures and seldom changes. On beaches with no consolidated supralittoral the limit was established where vegetation fully covered the sand. This polygon establishes the maximum landward limit of the beach, therefore has to be adjusted to the oldest reference in the database, and should be modified if alterations of the landward limit happen, such as those generated by constructions (Fig 3).

2. Image composite. Landsat images were converted to top of the atmosphere (TOA) reflectance and a median based composite was constructed from non-cloudy pixels applying the Landsat Simple Composite built-in function of GEE using all available bands. This function computes a Landsat composite from a collection of raw Landsat scenes, it applies a standard TOA calibration and assigns a cloud percentage to each pixel on each image. Then, a median-based value per band is computed for every pixel with a range of clouds lower than the selected cloud score range (CSR), generating the composite. The CSR was used to control the percentage of clouds per pixel, adapting the value until a clear true color visualization based on blue, red and green bands, was obtained (Fig 3).

For the Montevideo coast, beach area was estimated yearly using all the available standard Level 1 Terrain-corrected orthorectified images from Landsat 5 (L5) (1984–2011) and Landsat 7 (L7) (2004–2019). L5 had an average of 15 images per year with a minimum of 6 (1985 and 1985) and a maximum of 31 (2004), while L7 had an average of 33 images per year with a minimum of 28 (2010) and a maximum of 41 (2019) (Table 1). For overlapping years (2004-2011) areas were measured through both Landsat collections and averaged.

3. Water mask. In order to reduce the area of classification and limit the number of categories, the water layer was removed by applying a normalized difference water index (NDWI)

(Gao, 1996) mask with different water thresholds (WTH), combined with photo interpretation. This step does not seek to set the coastline, so a buffer zone of water-sand interface must remain for the algorithm to classify the cover.

4. Sand/Other training areas. The spectral signal at training areas acts as seed information for the developing of the Random Forest classification algorithm. For the sand category, training polygons were distributed along the coast covering the range of sand colors present on each composite. The category "other" was formed by polygons that comprised all non-sand cover, including urban, rocky and rural cover. Random selection of sampling points within training areas was implemented to reduce spatial autocorrelation and classification errors (Millard and Richardson, 2015), considering that the number of training points should be at least 10 times the number of variables used in the classification (Jensen, 2005).

For the Montevideo coast the 7(8) spectral bands of Landsat 5(7) were considered as classification variables, 100 training points were randomly extracted from the training areas of each category.

5. Sand classification. Random Forest procedure was applied due to its very high classification accuracy (Cutler *et al.*, 2007). This algorithm builds several classification trees and then decides each pixel class by the majority vote of all trees (Breiman, 2001). To provide an adjustment for the sand classification, different threshold of the majority vote (CTH) were tested. All pixels within the beach polygons with a vote greater than the CTH were considered as sand.

Following this procedure, 1000 decision trees were constructed for each year. For the Montevideo coast, sand area estimations were made with CTH of 0.35, 0.50 and 0.65 in order to try different decision thresholds.

6. Beach vegetation. To account for vegetation growing at the beach, all pixels within the beach polygons classified as "other" and with a normalized difference vegetation index (NDVI) greater than the vegetation threshold (VTH) were considered as beach vegetation. Beach area was estimated adding sand area and vegetation area within each beach polygon.

Estimations of beach vegetation area were taken with VTH of 0.15, 0.25, and 0.35 in order to try different vegetation thresholds.

7. Photo interpretation. The coherence of the classification and the adequacy of the beach polygon were visually analyzed comparing the spatially explicit results to the original composite (Fig. 3).

8. Calibration. To approximate the best model parameters (CTH and VTH), estimated beach area was compared to independent measures, considering the possible sources of error due to the different approaches.

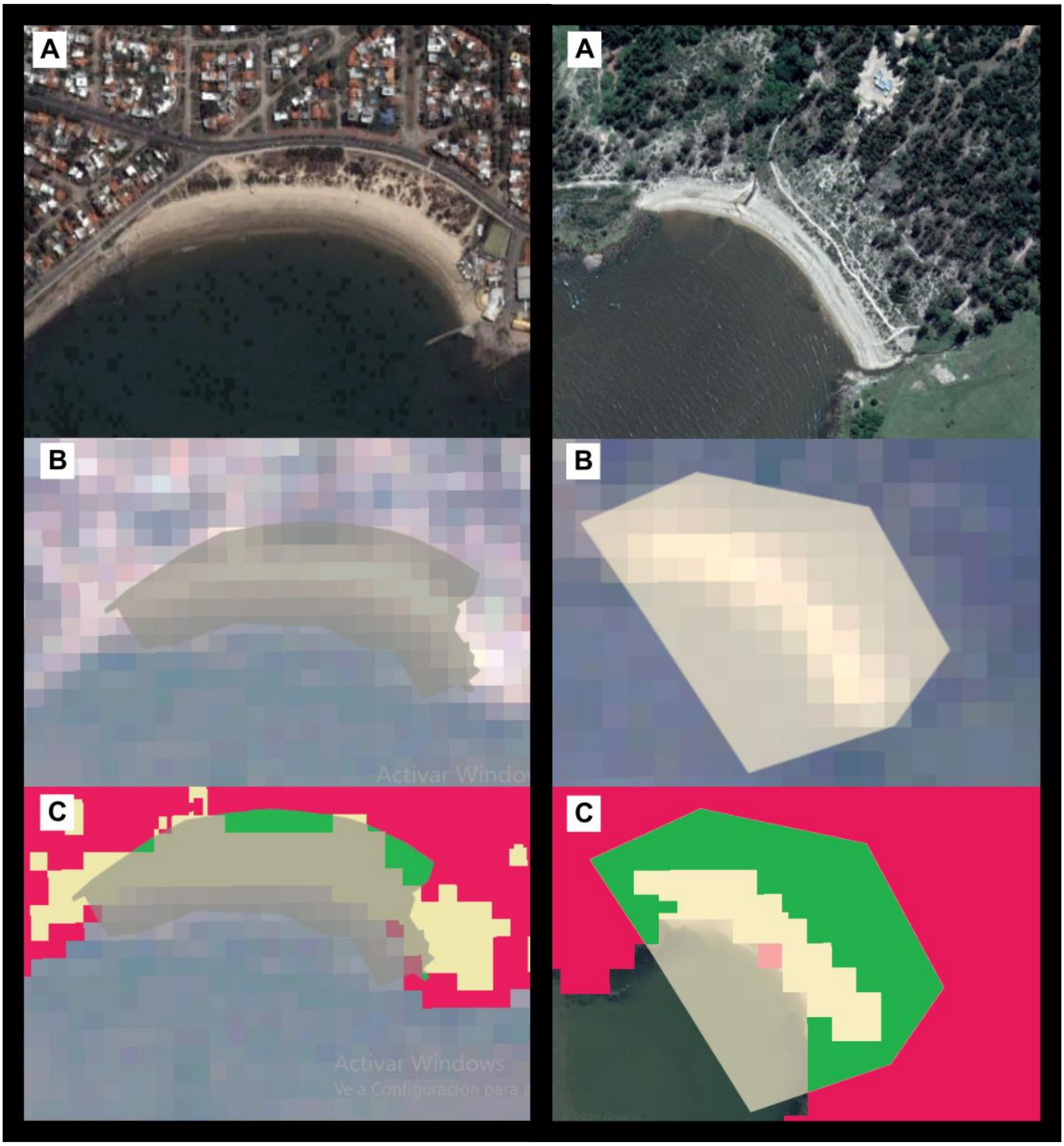

Figure 3. Diagram showing examples of the classification procedure for an urban beach (Verde) on the left and a beach with no with no consolidated supralittoral (Rocha). Panel A shows the Google Earth high-resolution imagery (July 9 of 2009), which was used as reference to construct beach polygons. B) Shows two inputs for the classification procedure, a 30m resolution Landsat 7 composite of the year 2009 with the geo-located Beach-polygon superimposed. C) Shows the classification output with yellow pixels indicating sand, red indicating "other" and green, beach-vegetation. The total area of the beach is equal to the areas in yellow and green under the polygon.

Table 1. Characteristics of the image collections used for the construction of the Landsat 5 and 7 composites to estimate beach area on the Montevideo coast, the number of images (n) is informed and the minimum and maximum date are provided, as day/month, for each year.

	Landsat 5			**Landsat 7**		
Year	**n**	**Min. date**	**Max. date**	**n**	**Min. date**	**Max. date**
1984	9	28/8	25/12	0	---	---
1985	6	10/1	5/12	0	---	---
1986	17	6/1	24/12	0	---	---
1987	13	9/1	18/12	0	---	---
1988	16	3/1	29/12	0	---	---
1989	6	14/1	22/5	0	---	---
1990	19	1/1	19/12	0	---	---
1991	12	4/1	6/12	0	---	---
1992	10	11/3	8/12	0	---	---
1993	17	9/1	27/12	0	---	---
1994	16	3/1	5/12	0	---	---
1995	16	15/1	17/12	0	---	---
1996	18	2/1	19/12	0	---	---
1997	8	4/1	20/11	0	---	---
1998	15	28/3	25/12	0	---	---
1999	13	1/1	12/12	0	---	---
2000	10	29/1	28/11	0	---	---
2001	14	31/1	17/12	0	---	---
2002	10	2/1	21/8	0	---	---
2003	13	16/7	23/12	0	---	---
2004	31	8/1	25/12	31	7/1	24/12
2005	27	1/1	26/11	32	2/1	27/12
2006	30	29/1	15/12	31	5/1	30/12
2007	19	16/1	29/9	32	8/1	17/12
2008	15	19/1	27/12	34	2/1	28/12
2009	23	5/1	7/12	29	4/1	22/12
2010	15	15/1	26/12	28	7/1	25/12
2011	16	18/1	2/11	38	3/1	28/12
2012	0	---	---	31	6/1	30/12
2013	0	---	---	33	15/1	24/11
2014	0	---	---	35	11/1	29/12
2015	0	---	---	36	5/1	16/12
2016	0	---	---	28	17/1	18/12
2017	0	---	---	38	10/1	28/12

2018	0	---	---	32	6/1	24/12
2019	0	---	---	41	16/1	27/12

Gutiérrez (2010) and Gutiérrez *et al.* (2016) present area measurements for 5 of the 20 beaches covered in this study (Ramírez, Pocitos, Buceo, Malvín, Brava). This independent information was obtained through manual measuring of satellite and aerial imagery covering a larger time period than the analyzed here, including two different methodologies for beach measuring, the previous high tide high water level (PHTH) (Buceo, Malvín and Brava) and the wet/dry line or run-up maxima (WDL) (Ramírez and Pocitos) (Boak and Turner, 2005). A total of 39 independent measures overlap with the period and sites covered in this study and were compared to the corresponding yearly total area estimate obtained using this methodology. Despite the differences in the acquisition procedures the independent information served as reference to calibrate the method parameters for the Montevideo coast.

Data analysis and results plot were performed using R software (R Development Core Team, 2012). Wilcoxon's Signed-Rank Test and Pearson's correlation coefficient were applied to analyze differences between independent and estimated data without assuming normality, estimations were compared with and without discriminating by the methodology of the independent information. Using the best performing classification parameters, total beach area was estimated for 20 beaches on the Montevideo coast, results were standardized to facilitate visualization and allow comparisons of area variations between beaches of different size and make evaluations at coastal level. The standard score for a given beach is obtained by subtracting the mean beach area in the entire period to the area on a year and dividing by the standard deviation for the whole period (standard score $beach_{(Y)}$ = ($area_{(Y)}$ - mean area)/standard deviation, where Y is the year) (Fig. 5).

RESULTS

Wilcoxon's paired test did not reject the equivalence of the estimated and independent medians of sandy beach area when it was performed with CTH=0.50 (Fig. 4). Correlation between the two approximations was high, with a maximum Pearson's score of 0.80, corresponding to CTH=0.50 and VTH=0.15 (Fig. 4).

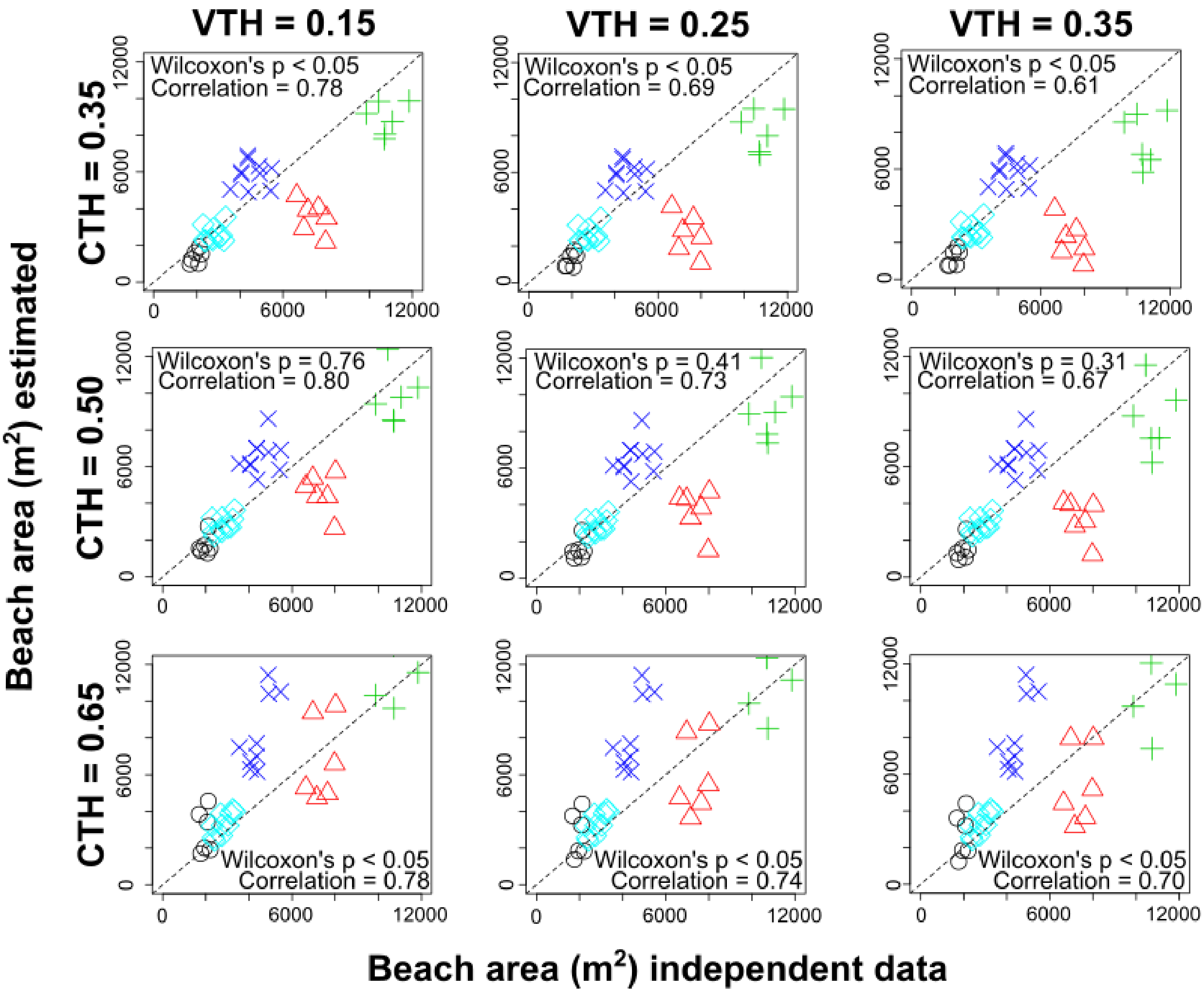

Figure 4. Scatter plots of the calibration for Beach area estimated considering different classification thresholds (CTH) and vegetation thresholds (VTH). At each scatterplot, the p-value of the Wilcoxon's Signed-Rank Test and the Pearson's coefficient of correlation with paired samples are shown. Each of the compared beaches is shown with a different shape: circles (black) correspond to Brava, rotated squares (sky-blue) to Ramírez, exes (blue) to Pocitos, triangles (red) to Buceo, crosses (green) to Malvín.

When independent measures were discriminated by measuring methodology an interesting pattern emerged, PHTH estimations were better correlated to measures obtained with CTH = 0.65 (Table 2). While WDL was non-discriminated from measures with CTH = 0.35 (Table 3). No major differences associated to the different VTH were registered at this stage of the analysis.

Table 2. Comparison of estimated areas with independent measures taken with the previous high tide high water level (PHTH) methodology. Wilcoxon's p-value and Pearson's correlation index are given for the comparisons with combinations of vegetation threshold (VTH) and classification threshold (CTH) values.

PHTH	**VTH = 0.15**	**VTH = 0.25**	**VTH = 0.35**
CTH = 0.35	Wilcoxon's p = 7.63e-06 Correlation = 0.90	Wilcoxon's p = 7.63e-06 Correlation = 0.85	Wilcoxon's p = 7.63e-06 Correlation = 0.81
CTH = 0.50	Wilcoxon's p = 0.0016 Correlation = 0.90	Wilcoxon's p = 0.00042 Correlation = 0.87	Wilcoxon's p = 0.00033 Correlation = 0.83
CTH = 0.65	Wilcoxon's p = 0.30 Correlation = 0.88	Wilcoxon's p = 0.87 Correlation = 0.86	Wilcoxon's p = 0.64 Correlation = 0.84

Table 3. Comparison of estimated areas with independent measures taken with the wet/dry line (WDL) methodology. Wilcoxon's p-value and Pearson's correlation index are given for the comparisons with different combinations of vegetation threshold (VTH) and classification threshold (CTH) values.

WDL	VTH = 0.15	VTH = 0.25	VTH = 0.35
CTH = 0.35	Wilcoxon's p = 0.058 Correlation = 0.84	Wilcoxon's p = 0.064 Correlation = 0.84	Wilcoxon's p = 0.064 Correlation = 0.84
CTH = 0.50	Wilcoxon's p = 0.0032 Correlation = 0.87	Wilcoxon's p = 0.0032 Correlation = 0.87	Wilcoxon's p = 0.0031 Correlation = 0.87
CTH = 0.65	Wilcoxon's p = 5.72e-06 Correlation = 0.93	Wilcoxon's p = 5.72e-06 Correlation = 0.93	Wilcoxon's p = 5.726e-06 Correlation = 0.93

The best adjustment on pooled data were obtained with CTH =0.50 and VTH = 0.15. These parameters showed high p-value and the best correlation with the independent information. Yearly sand and vegetation estimation for 20 beaches on the Montevideo coast for the period 1984-2016 were obtained applying these values (Fig. 5), the resulting database and the GEE scripts are openly available at http://doi.org/10.5281/zenodo.4327667 (Orlando, 2020). The pattern arising on standardized beach area (Fig. 5B) shows a common behavior, among sites, of beach area variations for the Montevideo coast.

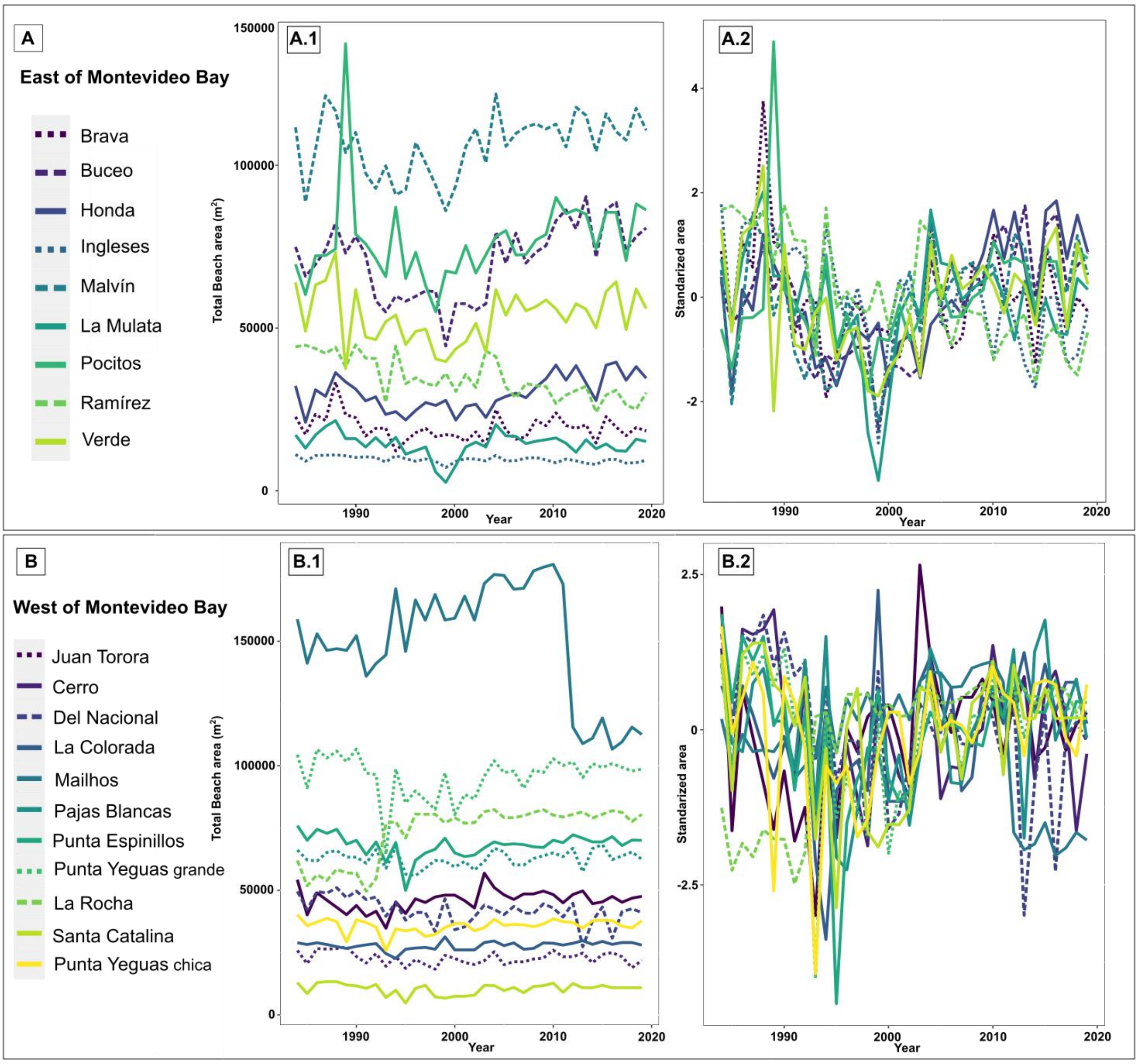

Figure 5. Beach area estimations for 20 sites analyzed in the 1984-2019 period. For visualization purposes beaches were divided geographically: Panel A shows information for beaches 12 to 20, located at the east of Montevideo Bay; Panel B shows beaches 1 to 11, located west of Montevideo Bay. In A.1 and B.1 the total beach area in square meters is shown, A.2 and B.2 show the standardized beach area variation for each portion of the coast.

DISCUSSION

This study describes and validates a methodology for estimating beach area, allowing single site studies as well as large scale coastal monitoring and the reconstruction of past information at a low operative cost. Mitigation of climate change effects and the sustainable use of sandy beach ecosystems is currently constrained by the available information (Harley *et al.*, 2010). The total area estimations obtained through the methodology presented allows to analyze individual beaches, while standardized values are useful to analyze coastal scale dynamics. The resulting database of beach area variation could allow a better, more informed, coastal management by: 1) fostering an improved understanding of the relationship between climate drivers and coastal change; 2) allowing quantified estimations of landscape attributes such as vegetation; 3) estimating recreational carrying capacity and 4) detecting erosive patterns. Although this methodology is unable to discriminate exotic and native vegetation, the evolution of vegetation area could be used to detect growth patterns associated with invasive exotic species.

The proposed methodology produces a yearly measure that reduces variability due to the time of the year or short-term climatic conditions by integrating year-round variations into a single image. This, together with the time span of the Landsat collection, makes this approach suitable as a standardized source of information for time series analysis (Orlando *et al.*, 2019). No significant differences were found between independent area measurements and estimations with CTH=0.5, even though the independent information (Gutiérrez *et al.*, 2016) was based on snapshots taken on different times of the year and associated with good weather conditions (aerial photography or single satellite pictures). Regarding water and sand boundaries the results of the proposed methodology suggest that the automated method gives an intermediate measure between WDL and PHTH techniques, WDL estimates a higher beach area as it depends on the water sand interface at the current time, while PHTH relies on the previous high tide. In Figure 3, the independent WDL measures of beaches are above the 1:1 line, while PHTH measures are below, a pattern supported by the results of Wilcoxon's p-value and Pearson's correlation. However, due to the spatial resolution of the Landsat series fine scale features such as this should be addressed carefully, the pattern found here could be influenced by tide range and other local conditions. The relationship between this methodology and the WDL and PHTH techniques should be analyzed at different locations in order to adequately assess correlations.

When standardized, all beaches on the Montevideo coast showed a common pattern of area variation, which has been related to variations on the climatic conditions (Orlando *et al.*, 2019). Further subdivision according to geographic position showed similar but different behaviors for Montevideo west and east coast. This level of analysis allows the detection of erosive phases as well as providing valuable input for large-scale (spatial and temporal) coastal planning.

The methodology presented here provides the necessary basis for quantifying sandy beach area, a major variable of sandy beach management and ecology. Furthermore, the temporal span of the Landsat collection allows to reconstruct beach area for over 30 years, this amount of previously unavailable information has the potential of improving knowledge on coastal dynamics and planning. In summary, the methodology presented and validated here can be easily applied anywhere, allowing an increase on quality and quantity of the information critically needed for coastal management at a low operational cost.

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