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POSGRADO EN CIENCIAS DEL MAR Y LIMNOLOGÍA**

**Conectando el pasado con el presente: evaluación de los cambios a largo plazo en la abundancia de la tortuga prieta, *Chelonia mydas*, en el Golfo de California**

**TESIS  
(POR ARTÍCULOS CIENTÍFICOS)**

QUE PARA OPTAR POR EL GRADO DE:

**DOCTORA EN CIENCIAS**

PRESENTA:

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## RESUMEN

La conservación exitosa de taxones longevos requiere de un entendimiento sólido de las líneas de base históricas y las tendencias poblacionales a largo plazo. El conocimiento ecológico local (CEL) puede contribuir a las prácticas de conservación y manejo al establecer líneas de base locales y metas de recuperación, evaluar los niveles de población local, integrar las dimensiones culturales de la interacción humano-ambiente y fomentar la generación de resultados sólidos. Proponemos un marco robusto para documentar y cuantificar el CEL, utilizando una agregación de forrajeo de tortuga prieta (*Chelonia mydas*) en Baja California, México como estudio de caso. Esta población altamente explotada de tortuga prieta llevó al borde de la extinción debido una pesquería comercial escasamente regulada que operó entre inicios de la década de 1950 y la década de 1980. No se disponían de datos técnicos de línea base de abundancia para este período, ya que los estudios biológicos comenzaron tras el colapso poblacional en 1995. Para llenar este vacío de datos, generamos una serie de tiempo de captura por unidad de esfuerzo (CPUE) estandarizada, derivada de CEL, que luego se integró con datos de monitoreo científico. Los modelos ajustados al CEL y a los datos de monitoreo (i) establecieron una línea base de abundancia; (ii) describieron los declives históricos, revelando que el declive más crítico (exponencial) ocurrió entre 1960 y 1980; y (iii) indicaron un aumento en la abundancia de tortuga prieta, impulsada por el reclutamiento de juveniles, pero solo tras de ~ 40 años de protección de los hábitats de anidación y alimentación de la especie en México. No obstante, de acuerdo con los datos obtenidos, la disminución de la población se produjo un 75% más rápido que el aumento, y la abundancia actual se encuentra en ~60% de los niveles de referencia históricos, lo que indica que aún es

necesario continuar con acciones de conservación a largo plazo. Este estudio demuestra el potencial de vincular CEL y la ciencia ecológica para proporcionar información crítica para la conservación y promoviendo, a la vez, futuros sostenibles para las comunidades locales.

## ABSTRACT

Successful conservation of long-lived taxa requires a reliable understanding of historical baselines and long-term population trends. Local Ecological Knowledge (LEK) can contribute to sound conservation and management practices by establishing local baselines and recovery targets, evaluating local population levels, integrating cultural dimensions of human-environment interaction, and supporting equitable outcomes. We propose a robust framework to document and quantify LEK, using a green turtle (*Chelonia mydas*) foraging aggregation in Baja California, Mexico as case study. This heavily exploited green turtle population was driven to near extinction by a largely unregulated commercial fishery from the early 1950s to the 1980s. No scientific baseline abundance data were available for this timeframe, as biological surveys began after population collapse in 1995. To fill this data gap, we generated a standardized, LEK-derived catch-per-unit-effort (CPUE) time-series which we then integrated with scientific monitoring data. Models fitted to LEK and monitoring data (i) established baseline abundance; (ii) described historical declines, revealing that the most critical (exponential) decline occurred between 1960 and 1980; and (iii) indicated increasing green turtle abundance driven by juvenile recruitment, but only after ~40 years of safeguarding the species' nesting and foraging habitats in Mexico. Nonetheless, population declines occurred 75% faster than increases, and current abundance is at ~60% of historical baseline levels, indicating that sustained, long-term conservation actions are still necessary. This study demonstrates the potential of linking LEK and ecological science to provide critical information for conservation while promoting sustainable futures for local communities.

## INTRODUCCIÓN

La conservación y el manejo de especies longevas y altamente migratorias —como las tortugas marinas— requiere de una amplia perspectiva espaciotemporal (Akçakaya et al., 2018; Lotze & Worm, 2009; Vierros et al., 2020). Por ejemplo, para evaluar el estado de conservación de una de estas especies, la Unión Internacional para la Conservación de la Naturaleza (IUCN, por sus siglas en inglés) exige datos que abarquen al menos tres generaciones (Harris et al., 2012; IUCN, 2018). Para especies longevas como las tortugas marinas, tres generaciones pueden abarcar más de 90 años, generando retos importantes en contextos donde no existen datos técnicos de monitoreo o línea base (McClenachan et al., 2016; Pauly, 1995; Sáenz-Arroyo et al., 2006). Algunos ejemplos pueden incluir las especies impactadas por pesquerías ilegales, escasamente reguladas o sin documentación, las pesquerías de pequeña escala o la cacería de subsistencia (Lee et al., 2018; Sáenz-Arroyo et al., 2005; Selgrath et al., 2018).

Ante los escenarios de datos técnicos escasos, el conocimiento ecológico local (CEL) constituye una valiosa fuente de información que puede contribuir al desarrollo de estrategias robustas de conservación y manejo, a la vez que fomenta la integración de las dimensiones culturales de las interacciones humano-ambiente y abona a la prácticas equitativas e incluyentes de co-construcción del conocimiento (Barrios-Garrido et al., 2018; Lee et al., 2018; Poe et al., 2014). El CEL se puede definir como conocimiento empírico geográficamente situado, adquirido por un grupo social específico respecto al ambiente y biota que lo rodea (cf. Narchi et al., 2014). El CEL, en conjunto con el acervo histórico, resulta particularmente importante en el caso de las tortugas marinas, y en particular la tortuga verde o prieta (*Chelonia mydas*). Esta especie ha estado sujeta a la explotación

humana durante siglos, y se cuenta con registros históricos y CEL detallado respecto a sus capturas y abundancia pasada (Early-Capistrán et al., 2018; McClenachan et al., 2016; Sáenz-Arroyo et al., 2006). En contraste, los datos de monitoreo científico de *C. mydas* en algunas regiones, como el Pacífico oriental, abarcan menos de 50 años (Balazs & Chaloupka, 2004; Márquez, 1996; Seminoff et al., 2015).

La tortuga prieta es un segmento poblacional distintivo del Pacífico oriental de la especie cosmopolita *C. mydas* (Dutton et al., 2019; Seminoff et al., 2015; Wallace et al., 2010). Si bien se le ha aplicado estatus específico o subespecífico a esta población (*C. agassizii*), los análisis genéticos no apoyan dicha distinción (Dutton et al., 2019; Karl et al., 1992; Seminoff et al., 2015), por lo que en este documento se utilizará el nombre científico *C. mydas*. Asimismo, el nombre común “tortuga prieta”, ampliamente usado en el Pacífico oriental, servirá para distinguir a este segmento poblacional de otras poblaciones de tortuga verde. Esta especie principalmente herbívora tiene un papel clave en la salud y estabilidad de las dinámicas de las redes tróficas marinas, así como la viabilidad y resiliencia a largo plazo de los ecosistemas marinos (Chaloupka et al., 2008; Heithaus et al., 2014; Jackson et al., 2001).

La evaluación del estado de conservación de esta especie es un reto particular, dada la complejidad espaciotemporal de su historia de vida. Es altamente fecunda, pero debido a que es longeva y de lenta maduración —podría tardar entre 17 y 50 años en alcanzar la edad reproductiva— es vulnerable a la actividad humana en todas sus etapas de vida (Seminoff et al., 2015; Turner Tomaszewicz et al., 2018; Van Houtan et al., 2014). Asimismo, es altamente migratoria y ocupa una gama de hábitats ampliamente distribuidos durante diferentes estadios de vida, principalmente zonas de alimentación cálido-templadas y zonas

de reproducción tropicales. Estos hábitats pueden estar separados por cientos o miles de kilómetros, y frecuentemente se encuentran en diferentes países (Godley et al., 2010; Seminoff et al., 2015). Cabe destacar que manifiesta una alta fidelidad por sus sitios de reproducción (filopatría) y los hábitats finales de alimentación (Broderick et al., 2007; Seminoff et al., 2015).

Hay fundamentalmente dos poblaciones de esta especie que anidan en el Pacífico mexicano: (a) en las costas de Michoacán y (b) en el archipiélago de Revillagigedo— pero que pasan su etapa juvenil y parte de su etapa adulta en sitios de alimentación subtropicales y templados del Noroeste de México (Delgado-Trejo & Alvarado Díaz, 2012; Dutton et al., 2014; Seminoff et al., 2008). Las poblaciones de *C. mydas* en el Pacífico mexicano y el Golfo de California tuvieron un marcado declive entre la década de 1960 y 1990 debido a la sobreexplotación comercial, que abarcó la sobrepesca en los hábitats de alimentación del noroeste de México y la cosecha de huevo en las costas de Michoacán (Delgado-Trejo & Alvarado Díaz, 2012; Early-Capistrán et al., 2018; Felger et al., 2005). Actualmente está clasificada como “En Peligro de Extinción” por la IUCN y en la legislación mexicana, y toda las especies de tortuga marina en México se encuentran en veda total y permanente desde 1990 (IUCN, 2018; NOM-059-SEMARNAT-2010, 2010).

Debe tomarse en cuenta que, a nivel mundial, los programas de monitoreo de tortugas marinas están fuertemente sesgados hacia las playas de anidación, en donde únicamente se cuantifican las hembras anidantes y la producción de nidos y huevos (cf. Seminoff & Shanker, 2008). No obstante, *C. mydas* pasa la mayor parte de su vida en los hábitats de forrajeo. Como juveniles, ocupan estos hábitats durante 20 años o más hasta alcanzar la madurez sexual y las tortugas adultas de ambos sexos los utilizan durante los periodos no-

reproductivos (Bjorndal et al., 2005; Godley et al., 2010; Seminoff et al., 2003). En contraste con las zonas de anidación en donde se tiene acceso solo a las hembras reproductoras, los hábitats de forrajeo brindan información demográfica acerca de una variedad de grupos etarios, aportando así perspectivas importantes respecto a las tendencias actuales y futuras de la abundancia poblacional (Seminoff et al., 2003).

En la actualidad, las poblaciones de *C. mydas* han aumentado significativamente en diversas regiones del mundo, incluyendo el Pacífico oriental y central, el Caribe, el Atlántico y la región Indo-Pacífico (Balazs & Chaloupka, 2004; Broderick et al., 2006; Chaloupka et al., 2008; Mazaris et al., 2017). No obstante, existen importantes huecos de información y, en la mayoría de los casos, no existen datos sólidos de línea base. Lo anterior es consecuencia directa de la trayectoria relativamente corta de monitoreo de tortugas marinas a nivel mundial (Chaloupka et al., 2008; McClenachan et al., 2006; Van Houtan & Kittinger, 2014). Este es también el caso de las poblaciones de *C. mydas* en el Golfo de California. Gracias a décadas de esfuerzos de conservación a lo largo del área de distribución de esta especie en México —desde la protección de la playa de Colola en 1979, la suspensión de actividades pesqueras de tortuga prieta en 1983 y la veda total en 1990— las poblaciones en sitios de alimentación y forrajeo han aumentado desde inicios de la década de 2000 (Delgado-Trejo & Alvarado Díaz, 2012; Seminoff et al., 2015). No obstante, se requiere de una mejor comprensión de su historial completo, sobre todo previo al colapso poblacional por la sobreexplotación comercial, para lograr una mejor comprensión de las interrelaciones entre las características de la dinámica poblacional de la especie y los cambios entre prácticas extractivas y posteriormente de conservación. Solo así se podrá lograr evaluar de manera robusta su estado de recuperación.



## **Sitio de estudio**

Bahía de los Ángeles, Baja California, México (28.951917°, -113.562433°), es un hábitat de alimentación cálido-templado en la región de las grandes islas del Golfo de California (Seminoff et al., 2003, 2020). Esta región se caracteriza por procesos intensos de surgencia y mezcla por efectos de marea, los cuales generan condiciones de alta productividad de comunidades bentónicas en las que predominan las algas marinas (Seminoff et al., 2003). Al realizar este estudio, la comunidad de Bahía de los Ángeles contaba con una población de 639 habitantes (INEGI, 2021). La economía local se basa principalmente en la pesca y el buceo comercial y —de manera secundaria— el turismo estacional, principalmente orientado a la pesca deportiva y el ecoturismo.

Bahía de los Ángeles (BLA) se considera un sitio índice de alimentación al contar con (i) agregaciones abundantes y recurrentes de tortugas en el ambiente costero-nerítico y (ii) monitoreo sistemático y constante (>10 años), por lo que se puede suponer que la información obtenida refleja patrones generales existentes en otros sitios de la región utilizados por la misma población (Early-Capistrán et al., 2020). Entre 1960 y 1980, BLA fue uno de los sitios más importantes y productivos de la pesquería comercial de tortuga prieta en el Pacífico oriental (Caldwell, 1963; Early-Capistrán et al., 2018). Los esfuerzos de monitoreo y conservación en la localidad iniciaron a finales de la década de 1970 a través de programas gubernamentales (Seminoff et al., 2008). El monitoreo en agua comenzó en 1995, y continúa hasta la fecha gracias a esfuerzos conjuntos de instancias gubernamentales, académicas y de la sociedad civil (Seminoff et al., 2008).

Al igual que otros sitios de forrajeo, la población de BLA está compuesta por individuos de los dos *stocks* genéticos regionales, Michoacán y Revillagigedo. No obstante, está poco establecida la composición y el aporte proporcional de los diferentes *stocks* genéticos en las agregaciones de tortugas que se alimentan en Bahía de los Ángeles (Dutton et al., 2019; Seminoff et al., 2015). No hay actividad de anidación en BLA. No obstante, se conoce que la mayoría (56-71%) de los individuos en las agregaciones en zonas de forrajeo del Golfo de California provienen de la población anidadora de Colola, Michoacán (~1500 km al sureste de BLA, 18.297392°, -103.410956°) (Koch, 2013). Asimismo, la población que anida en Colola es la de mayor importancia de *C. mydas* en el Pacífico oriental con monitoreo a largo plazo (>30 años), por lo que se considera un sitio índice de anidación que sirve como referencia confiable para las poblaciones que ocupan el hábitat de alimentación de BLA (Delgado-Trejo & Alvarado Díaz, 2012; Seminoff et al., 2003).

### **Reconstrucción y evaluación de la abundancia**

Presentamos un estudio de caso de la tortuga prieta en el Golfo de California en el que se realiza una aproximación transdisciplinaria hacia la ciencia de la conservación, integrando el CEL con datos de monitoreo biológico proporcionados por la Comisión Nacional de Áreas Naturales Protegidas (CONANP), el Grupo Tortuguero de Bahía de los Ángeles y el Dr. Jeffrey Seminoff (NOAA). A través de este estudio de caso, demostramos una metodología novedosa que puede adaptarse a otras especies longevas y fuertemente explotadas para evaluar las tendencias a largo plazo en escenarios de datos técnicos escasos. La presente obra está compuesta de dos artículos científicos, presentes a modo de capítulos, seguidos de una

sección de discusión general y de conclusiones generales, donde ambos artículos se abordan en conjunto.

El primer artículo, publicado en la revista *PeerJ* (Early-Capistrán et al., 2020), abarca la reconstrucción de la captura-por-unidad-de-esfuerzo (CPUE) desde inicios de la pesquería comercial de *C. mydas* en la zona de estudio a inicios de la década de 1950 hasta su colapso comercial a inicios de la década de 1980. Esta reconstrucción permitió (i) definir una línea base de abundancia previa a la explotación comercial a gran escala; (ii) establecer metas de recuperación y (iii) generar datos estandarizados del historial de largo plazo que pudieran integrarse directamente con los datos recientes de monitoreo científico. La realización de este artículo conllevó el desarrollo de una metodología novedosa para cuantificar el CEL, a través de un proceso de documentación, corroboración y clasificación del CEL a través de la etnografía; síntesis del CEL a través de la codificación, indexación. Asimismo, se integraron procesos de retroalimentación con análisis estadísticos (Early-Capistrán et al., 2020).

El segundo artículo, publicado como *pre-print* en el repositorio *bioRxiv*, abarca la integración de los datos derivados del CEL con datos de monitoreo para generar, hasta donde se conoce, la serie de tiempo de abundancia estandarizada más extensa a nivel mundial para una zona de alimentación de tortuga marina (1952-1982). Usando novedosos métodos de imputación múltiple por ecuaciones en cadena (MICE, por sus siglas en inglés) para suplir los datos faltantes, se evaluaron los cambios a largo plazo y la abundancia actual de la población local Bahía de los Ángeles en relación con los niveles de línea base histórica derivados del CEL (Early-Capistrán et al., 2021).

En este trabajo, se busca entender el cambio a largo plazo de *C. mydas* en un sitio índice de alimentación y contextualizar su nivel de abundancia actual con base en una línea base histórica robusta. Este proceso ayudará a fundamentar procesos y estrategias de conservación y manejo, así como líneas de investigación futuras. Además, esta visión integrativa del CEL con el monitoreo ecológico contribuye a la sinergia de la capacidad colectiva de las comunidades locales y la comunidad científica, aportando a nuevos procesos de conservación más justos e incluyentes (Game et al., 2015).

## ANTECEDENTES

Este trabajo abarca tres etapas fundamentales. Primero, se documentó, procesó y cotejó CEL en la comunidad de los Ángeles. Segundo, se desarrolló un método novedoso para reconstruir series de tiempo estandarizadas de captura por unidad de esfuerzo de *C. mydas* a partir del CEL. Este proceso permitió reconstruir la abundancia local de esta especie en Bahía de los Ángeles durante la explotación comercial y previo al monitoreo científico. Tercero, esta serie de tiempo se integró con los datos recientes de monitoreo científico para modelar los cambios a largo plazo en la abundancia local, y así evaluar de manera robusta el estado de conservación de la población local y generar nuevas perspectivas acerca de la dinámica poblacional. A continuación, se presentan los antecedentes principales de este estudio de caso.

## **La tortuga prieta en Bahía de los Ángeles**

Bahía de los Ángeles es un sitio destacado para el estudio de *C. mydas* tanto en términos biológicos como culturales (Early-Capistrán et al., 2018; Seminoff et al., 2003, 2008). La tortuga prieta es una especie de gran importancia biocultural en la península de Baja California, y ha tenido roles fundamentales como fuente de alimento, medicina y materiales durante todas las etapas de ocupación humana en la región (Averett, 1920; Caldwell, 1962; Early-Capistrán et al., 2018; Garibaldi & Turner, 2004; Ritter, 2010). *C. mydas* aparece en el registro arqueológico desde las primeras etapas de ocupación humana, hace al menos 12,000 años y fue una fuente importante de proteína para los pueblos yumanos hasta inicios de la colonización a finales del siglo XVII (Bowen et al., 2008; Des Lauriers, 2011; Laylander, 2010). Durante la ocupación colonial y poscolonial, la tortuga prieta mantuvo su importancia como fuente de alimentación en las rancharías y comunidades mestizas (cf. Early-Capistrán, 2014; Early-Capistrán et al., 2018). En contraste con otras regiones del mundo —como el Caribe o el Golfo de México— donde las poblaciones de *C. mydas* colapsaron al poco tiempo de la colonización europea y la integración a los mercados globales (Jackson et al., 2001; McClenachan et al., 2006), en Baja California el aislamiento geográfico respecto a los mercados nacionales e internacionales, la pequeña población humana y las limitaciones tecnológicas de la pesca permitieron que las capturas se mantuvieran relativamente bajas y sustentables hasta mediados del siglo XX (Early-Capistrán et al., 2018).

La trayectoria singular de la explotación de *C. mydas* en Baja California permite situar una línea base histórica confiable dentro de la memoria intergeneracional de las comunidades pesqueras. A través de la investigación de maestría de la sustentante, enfocada en la ecología histórica marina y realizada en el PCMyL-UNAM bajo la dirección de la Dra. Andrea Sáenz-

Arroyo, se identificaron los inicios de la década de 1950 como una línea base histórica confiable para esta población de tortuga prieta (Early-Capistrán, 2014; Early-Capistrán et al., 2018). Hasta mediados del siglo XX, las tortugas se capturaban de manera sustentable utilizando arpones, en canoas impulsadas con remos. Destaca el efecto de la segunda guerra mundial en el desarrollo de nuevas tecnologías para la explotación del ambiente marino, que conllevó a la mayor tecnificación e industrialización de las pesquerías a nivel mundial (Longo et al., 2015). En el caso de la península de Baja California, la segunda guerra mundial y la posguerra destacan como un parteaguas en el desarrollo de pesquerías comerciales e industrializadas (Early-Capistrán, 2014; Shepard-Espinoza & Danemann, 2008).

Entre 1960 y 1980, el crecimiento de las ciudades fronterizas generó una gran demanda de carne y subproductos de tortuga prieta, que se surtía principalmente con tortugas de las importantes zonas de forrajeo del Pacífico y el Golfo de California; entre éstas, Bahía de los Ángeles destacaba por su alta productividad (Caldwell, 1962, 1963; Early-Capistrán et al., 2018; Felger et al., 1976; Seminoff et al., 2008). Esto coincidió con la llegada de tecnologías como los motores fuera de borda, las redes tortugueras, el monofilamento de *nylon* y las embarcaciones de fibra de vidrio que aumentaron la capacidad de captura y desplazamiento de la flota. A la vez, la construcción de la carretera transpeninsular a inicios de la década de 1970 facilitó el acceso a los mercados regionales y nacionales (Early-Capistrán et al., 2018). El conjunto de demanda, tecnología y vías de comunicación llevó a la especie al borde de la extinción (Early-Capistrán et al., 2018; Felger et al., 2005; Seminoff et al., 2008).

Los procesos de conservación e investigación a largo plazo en la comunidad iniciaron a finales de la década de 1970 en respuesta al declive de las poblaciones, mediante esfuerzos

gubernamentales impulsados por el oceanólogo Antonio Reséndiz. La pesquería colapsó a inicios de la década de 1980, y las licencias de captura de tortuga prieta se suspendieron en 1983 (Márquez, 1996). El monitoreo en agua comenzó en 1995, tras el colapso poblacional, y continúa hasta la actualidad gracias al esfuerzo conjunto de instancias gubernamentales, académicas y de la sociedad civil (Seminoff et al., 2008). En la actualidad, la población de tortuga prieta en Bahía de los Ángeles ha mostrado señales de aumento (Early-Capistrán et al., 2018; Seminoff et al., 2015). No obstante, la evaluación robusta de su estado de conservación requiere de una comparación de la abundancia actual con una línea base previa al colapso poblacional (McClenachan et al., 2006, 2012; Pauly, 1995).

### **El CEL y la ciencia de la conservación**

Dentro de la ciencia de la conservación, cada vez es más aparente la necesidad de integrar visiones a largo plazo, así como procesos socioculturales y socioambientales, para evaluar los estados de conservación de las especies y generar resultados más justos y equitativos (Barrios-Garrido et al., 2017; Lee et al., 2018; Mistry & Berardi, 2016). En este contexto, el uso de datos históricos, datos grises y CEL han cobrado una creciente importancia en la ciencia de la conservación (Lee et al., 2018; Poe et al., 2014; Selgrath et al., 2018). El CEL y el registro histórico se han utilizado en diversos contextos para evaluar cambios a largo plazo en la abundancia de especies explotadas, generar datos de línea base de abundancia y llenar huecos de información que no podrían abordarse si se utilizaran únicamente datos técnicos (Mason et al., 2019; Mukherjee et al., 2019; Revollo-Fernández & Sáenz-Arroyo, 2012; Sáenz-Arroyo et al., 2005, 2006). Algunos ejemplos incluyen la evaluación de los cambios ecológicos en grandes escalas de tiempo (Jackson, 2001; Lee et al., 2018; Lotze

et al., 2011, 2011; Sáenz-Arroyo et al., 2006); el desarrollo de planes colaborativos de conservación y manejo de especies amenazadas (Mason et al., 2020; Wedemeyer-Strombel et al., 2019); la documentación del patrimonio biocultural (Johannes, 1981; Malm, 2015; Si & Lahe-Deklin, 2015) o el desarrollo de marcos conceptuales para la modelación ecológica a partir de datos no-convencionales (Ainsworth, 2011; Bélisle et al., 2018). En el caso de las tortugas marinas, existen reconstrucciones de abundancia y distribución de anidación de tortuga verde a partir de datos históricos en el Caribe y Hawái (McClenachan et al., 2006; Van Houtan & Kittinger, 2014). Asimismo, existen importantes trabajos de evaluación del estado de conservación que integran aspectos socio-culturales de las tortugas en el Caribe, el Pacífico oriental y central y la región del Indo-Pacífico (Álvarez-Varas et al., 2020; Barrios-Garrido et al., 2020; Shanker & Kutty, 2005; Wedemeyer-Strombel et al., 2019).

### **Modelación ecológica a largo plazo**

La complejidad espaciotemporal de la historia de vida de las tortugas marinas limita las posibilidades de observación directa, ya que sus procesos biológicos y dinámica poblacional se desarrollan en escalas temporales de décadas (o incluso siglos al considerar múltiples generaciones), y a través de hábitats separados por cientos o miles de kilómetros, frecuentemente en diferentes países (Broderick et al., 2006; Godley et al., 2010; Seminoff & Shanker, 2008). Asimismo, el grueso de la investigación a nivel global se concentra en una sola fase de vida, con énfasis en las playas de anidación (Casale & Heppell, 2016; Hamann et al., 2010; Seminoff & Shanker, 2008). Existen importantes retos debido a la falta de parámetros básicos, incluyendo la edad de maduración, la duración de las etapas de vida y la conectividad demográfica y migratoria, entre otros (Casale & Heppell, 2016). Esto se debe,



en parte, a la falta de datos a largo plazo de diferentes etapas de vida y hábitats (Casale & Heppell, 2016; Seminoff & Shanker, 2008; Van Houtan et al., 2014). Por ende, la modelación matemática es una herramienta fundamental para evaluar la dinámica poblacional y los cambios de abundancia (Becker et al., 2019; Casale & Heppell, 2016; Chaloupka & Balazs, 2007; Piacenza et al., 2016, 2017).

En el caso de la tortuga verde, existen importantes estudios de modelación a largo plazo en Japón (Kondo et al., 2017), Hawái (Chaloupka et al., 2008; Chaloupka & Balazs, 2007; Piacenza et al., 2017), el Mediterráneo (Casale & Heppell, 2016) y Australia (Milani Chaloupka & Limpus, 2001; Laloë et al., 2020). Si bien la mayoría de los estudios y modelos a nivel mundial se han enfocado en las tendencias en zonas de anidación, existe un énfasis cada vez mayor en el desarrollo de modelos que evalúen la conectividad demográfica y entre hábitats (Casale & Heppell, 2016; Piacenza et al., 2016). Asimismo, ante el crecimiento poblacional aparente de poblaciones anidadoras —en el Pacífico central y oriental, la región Indo-Pacífico, el Atlántico y el Caribe— como resultado de aplicación prolongada de medidas de conservación, existe una necesidad creciente de evaluaciones de los cambios a largo plazo en la abundancia (Broderick et al., 2006; Chaloupka et al., 2008; Mazaris et al., 2017). Dado que en la mayoría de los casos solo existen datos técnicos a corto plazo —frecuentemente recopilados después del colapso poblacional debido a la sobreexplotación— se requiere de información sólida de línea base para evaluar, de manera robusta y fidedigna, el grado de recuperación de estas poblaciones (Kittinger et al., 2013; Mazaris et al., 2017). En el presente trabajo, se propone una metodología innovadora que permite generar dichos datos de línea base, combinarlos con datos de monitoreo científico actuales y contextualizar el estado poblacional dentro de una trayectoria a largo plazo. Esta aproximación podrá

modificarse y adaptarse para ser utilizada en otros sitios y con otras especies longevas, como los elasmobranquios, cetáceos, meros o sirénidos.

## JUSTIFICACIÓN

El monitoreo científico sistemático de *C. mydas* en Bahía de los Ángeles inició en 1995, tras la veda total de capturas en 1990 y el auge en la participación de organizaciones académicas, gubernamentales y de la sociedad civil en las tareas de investigación y conservación de las tortugas marinas (Seminoff et al., 2008). Para estas fechas, la población se encontraba mermada por décadas de explotación comercial a gran escala, por lo que limitarse a los datos de monitoreo conlleva un fuerte riesgo de hacer estimaciones sesgadas de la abundancia histórica y el estado de conservación actual (Pauly, 1995; Sáenz-Arroyo et al., 2005; Seminoff et al., 2008). En el presente trabajo, se busca contribuir a los aspectos teóricos y técnicos de la evaluación del estado de conservación de una especie longeva en un escenario de datos técnicos escasos.

Existen varios retos para evaluar el estado de conservación de la tortuga prieta: (i) la falta de monitoreo científico a largo plazo, que ha impedido determinar una línea base de abundancia confiable; (ii) la falta de datos empíricos de aspectos básicos de su biología y demografía, sobre todo para la etapa juvenil y la ocupación de zonas de forrajeo y (iii) la compleja historia de vida. Todo lo anterior dificulta el análisis holístico de la abundancia que integre diferentes etapas de vida y hábitats (Casale & Heppell, 2016; Seminoff & Shanker, 2008). Ante ello, se propone un enfoque novedoso para (a) crear estimaciones robustas de

abundancia a partir del CEL, (b) compararlos y contrastarlos con información reciente y (c) establecer una línea base para evaluar los cambios poblacionales a largo plazo y la abundancia actual a nivel local.

### **Aportes metodológicos**

Si bien existen avances importantes en el uso del CEL en la modelación ecológica y la ciencia de la conservación, es un campo naciente con potencial para aportar a los aspectos teórico-metodológicos, epistemológicos y prácticos de la incorporación de datos no-convencionales a la ecología marina (Bélisle et al., 2018; Moon et al., 2019; Young et al., 2018). En particular, se requieren marcos metodológicos para integrar, de manera robusta, el conocimiento empírico de las comunidades pesqueras con los marcos referenciales de la ecología. Asimismo, es necesario desarrollar aproximaciones interdisciplinarias para recopilar y cotejar información de diversas fuentes de conocimiento a partir de métodos disciplinarios sólidos, tanto de las ciencias sociales como de las ciencias naturales (St. John et al., 2014; Sutherland et al., 2014, 2018). La mayoría de los estudios realizados se han basado en la recopilación de anécdotas o cuestionarios, y la participación de científicos sociales frecuentemente se ha limitado a la verificación de datos numéricos y no a la generación de aproximaciones críticas a problemas teóricos y metodológicos fundamentales (Moon et al., 2019; St. John et al., 2014; Sutherland et al., 2018). En este trabajo, se presenta un marco metodológico flexible, de orientación transdisciplinaria, que integra métodos robustos de las ciencias naturales y sociales. Esta aproximación puede modificarse para generar estimaciones de abundancia de tortugas marinas y otras especies longevas con antecedentes de explotación humana prolongada (Early-Capistrán et al., 2020).

Cabe destacar que los datos faltantes o incompletos constituyen un problema recurrente en los trabajos de ecología histórica o las reconstrucciones basadas en datos grises o no-tradicionales (Early-Capistrán, 2014; Thurstan et al., 2014). Ante este escenario, se ha integrado el uso de las imputaciones múltiples por ecuaciones en cadena (MICE, por sus siglas en inglés), que utiliza Cadenas de Markov-Monte Carlo para generar múltiples series de datos y correr, paralelamente, múltiples ajustes de modelo que posteriormente se promedian tomando en cuenta la variabilidad entre las mismas (Nguyen et al., 2017; van Buuren, 2018). Esta herramienta sistemática para abordar los datos faltantes se ha utilizado ampliamente en la economía y las ciencias de la salud y el comportamiento, pero sus aplicaciones a la ecología aún han sido limitadas (Thurstan et al., 2014). El uso de MICE ha demostrado ser robusto en casos con hasta 75% de datos faltantes (Takahashi, 2017). En el presente trabajo se muestra una implementación innovadora de MICE al ser, hasta donde se sabe, el primer caso en donde se utilizan en conjunto con datos derivados del CEL.

### **Modelación ecológica aplicada de especies longevas**

La investigación a largo plazo es indispensable para entender las dinámicas ecológicas que ocurren en escalas de tiempo prolongadas (décadas o siglos), que no son visibles si los datos se limitan a lapsos temporales cortos (Giron-Nava et al., 2017; Jackson et al., 2001; Thurstan et al., 2014). Asimismo, el entendimiento de las dinámicas no-lineales y no-estacionarias — que son comunes en los procesos ecológicos— requiere series de largo plazo; el ajuste de modelos a series cortas conlleva limitaciones importantes para explicar los procesos de largo alcance (cf. Giron-Nava et al., 2017). Estas consideraciones cobran gran importancia en las especies longevas, cuyos procesos biológicos y dinámica poblacional se desarrollan en

escalas que van desde varias décadas hasta siglos, y se agudizan en aquellas que han sido sujetas a la explotación humana de manera intensiva o durante periodos prolongados (Chaloupka et al., 2008; Early-Capistrán et al., 2020; McClenachan et al., 2006). Los retos del cambio climático acrecientan la importancia de contar con modelos predictivos robustos generados a partir de datos de largo plazo y que integran múltiples fuentes de información, como el CEL y el registro histórico (Giron-Nava et al., 2017; Klein & Thurstan, 2016; Klenk & Meehan, 2015).

En el caso de las tortugas marinas, y de *C. mydas* en particular, la falta de datos a largo plazo de sitios de forrajeo —que incluyen tortugas juveniles y tortugas adultas de ambos sexos— ha limitado el entendimiento de la dinámica poblacional de la especie y la evaluación holística de su estado de conservación (Koch et al., 2007; Seminoff et al., 2003, 2015). La estimación de la abundancia pasada de *C. mydas* en Bahía de los Ángeles a partir del CEL, en conjunto con los datos de monitoreo científico, ha permitido generar una serie de tiempo de largo plazo (1952-2018). Esta serie de tiempo incluye una línea base histórica robusta que permite evaluar la abundancia actual y contextualizar el proceso de aumento poblacional observado en la localidad desde la década del 2000, contribuyendo de manera importante al desarrollo de estrategias de manejo y conservación y al conocimiento básico de la especie (Bjorndal et al., 2005; Seminoff et al., 2008).

## PREGUNTAS E HIPÓTESIS

### Preguntas de investigación

1. ¿Cuáles han sido la magnitud y los cambios en el tiempo de la abundancia de juveniles y adultos de *C. mydas* en Bahía de los Ángeles, desde de la pesquería comercial a gran escala (circa 1950) hasta el presente?
2. ¿Cuáles son los factores que mejor explican los patrones de variación a largo plazo en las tendencias de abundancia de *C. mydas* en Bahía de los Ángeles?
3. ¿Cuál es el estado de conservación actual de *C. mydas* en Bahía de los Ángeles, en comparación con los años anteriores a la pesca comercial a gran escala?

### Hipótesis

1. Durante la captura legal, la abundancia local de juveniles y adultos de *C. mydas* en Bahía de los Ángeles ha sido definida por la demanda comercial y la eficiencia pesquería durante los períodos de captura legal; durante las últimas tres décadas, ha sido definida por las medidas de conservación en combinación con el reclutamiento desde las zonas de anidación.
2. Las fluctuaciones de abundancia local de *C. mydas* en Bahía de los Ángeles son resultado principalmente de un incremento en el reclutamiento de juveniles, y que se relacionan como efecto de la veda total y de la introducción de protección de playas de anidación.
3. La tendencia a la baja de la abundancia de juveniles y adultos de *C. mydas* en Bahía de los Ángeles se ha revertido significativamente desde el inicio del monitoreo en agua hasta el presente, pero sigue significativamente por debajo de la línea base histórica.

## OBJETIVOS

### Objetivo general

Analizar las interrelaciones entre los patrones y factores de cambio a largo plazo en la abundancia local de *C. mydas* en un sitio índice de alimentación del Golfo de California (Bahía de los Ángeles, Baja California, BLA), con base en una serie de tiempo de abundancia local a largo plazo desarrollada a partir de datos históricos y etnográficos.

### Objetivos específicos

1. Desarrollar una metodología para generar series de tiempo de CPUE para *C. mydas* en un sitio índice de alimentación (BLA) a partir de datos históricos y etnográficos, con la cual construir una serie de tiempo (1950-2017) de abundancia local.
2. Desarrollar e implementar un modelo dinámico para analizar los cambios en la abundancia local de juveniles y adultos de *C. mydas* en BLA a partir de las series de tiempo generadas en el objetivo específico 1.

# **ARTÍCULO 1: CUANTIFICACIÓN DEL CONOCIMIENTO ECOLÓGICO LOCAL PARA MODELAR LA ABUNDANCIA HISTÓRICA DE FAUNA LONGEVA Y ALTAMENTE EXPLOTADA**

## **Resumen**

La obtención de tendencias poblacionales históricas robustas de especies longevas sujetas a explotación humana resulta desafiante en aquellos escenarios de datos escasos, como ocurre frecuentemente con especies afectadas por las pesquerías de pequeña escala o la cacería de subsistencia. Cada vez es más reconocido el valor del conocimiento ecológico local (CEL) en la conservación, tanto para descubrir tendencias históricas como para promover la gestión comunitaria de la información histórica. Proponemos un marco socioecológico mixto para documentar y cuantificar el CEL de manera confiable. Nuestro método puede ser adaptado por equipos interdisciplinarios para diversos taxones longevos con trayectorias de uso humano. Demostramos la validez de nuestro enfoque mediante la reconstrucción de datos de abundancia a largo plazo de la tortuga prieta (*Chelonia mydas*) en Baja California, México. Esta especie fuertemente explotada fue llevada al borde de la extinción la década de 1950 y la década de 1980 por una pesquería comercial escasamente reglamentada. No se disponían de datos científicos de línea base para este periodo, ya que los estudios biológicos comenzaron en 1995 tras del cierre de la pesquería. Para llenar este vacío de información, documentamos el CEL entre los pescadores locales utilizando métodos etnográficos. Luego, establecimos un marco iterativo para sintetizar y cuantificar el CEL utilizando modelos lineales generalizados (MGL) y regresión no lineal (RNL) para generar una serie de tiempo



estandarizada de captura por unidad de esfuerzo (CPUE) derivada de CEL. Verificamos la precisión de las estimaciones mediante comparaciones con estadísticas de pesca disponibles para 1962-1982 y modelamos las tendencias utilizando RNL. Nuestro modelo estableció una línea base de abundancia y describió los declives históricos, revelando que el declive más crítico ocurrió entre 1960 y 1980. Esta integración sólida de los datos de CEL con la ciencia ecológica contribuye a una visión holística del estado de conservación histórico y contemporáneo de esta especie.

## Sobretiro del artículo 1

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# Quantifying local ecological knowledge to model historical abundance of long-lived, heavily-exploited fauna

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## ABSTRACT

Deriving robust historical population trends for long-lived species subject to human exploitation is challenging in scenarios where long-term scientific data are scarce or unavailable, as often occurs for species affected by small-scale fisheries and subsistence hunting. The importance of Local Ecological Knowledge (LEK) in data-poor scenarios is increasingly recognized in conservation, both in terms of uncovering historical trends and for engaging community stewardship of historic information. Building on previous work in marine historical ecology and local ecological knowledge, we propose a mixed socio-ecological framework to reliably document and quantify LEK to reconstruct historical population trends. Our method can be adapted by interdisciplinary teams to study various long-lived taxa with a history of human use. We demonstrate the validity of our approach by reconstructing long-term abundance data for the heavily-exploited East Pacific green turtle (*Chelonia mydas*) in Baja California, Mexico, which was driven to near extinction by a largely unregulated fishery from the early 1950s to the 1980s. No scientific baseline abundance data were available for this time-frame because recent biological surveys started in 1995 after all green turtle fisheries in the area were closed. To fill this data gap, we documented LEK among local fishers using ethnographic methods and obtained verified, qualitative data to understand the socio-environmental complexity of the green turtle fishery. We then established an iterative framework to synthesize and quantify LEK using generalized linear models (GLMs) and nonlinear regression (NLR) to generate a standardized, LEK-derived catch-per-unit-effort (CPUE) time-series. CPUE is an index of abundance that is compatible with contemporary scientific survey data. We confirmed the accuracy of LEK-derived

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CPUE estimates via comparisons with fisheries statistics available for 1962–1982. We then modeled LEK-derived abundance trends prior to 1995 using NLR. Our model established baseline abundance and described historical declines, revealing that the most critical (exponential) decline occurred between 1960 and 1980. This robust integration of LEK data with ecological science is of critical value for conservation and management, as it contributes to a holistic view of a species' historic and contemporary conservation status.

**Subjects** Ecology, Marine Biology

**Keywords** Interdisciplinary studies, Ecological modelling, Local Ecological Knowledge (LEK), Sea turtles, Ethnobiology, Conservation, Long-lived fauna, Data-poor fisheries

## INTRODUCTION

Assessment of the current population status of long-lived species benefits from a firm understanding of historical baseline abundances (Pauly, 1995). For example, the International Union for Conservation of Nature (IUCN) Red List criteria requires abundance trends over three generations. For long-lived species, tracking three generations may necessitate >100 years of data (Seminoff & Shanker, 2008; IUCN, 2019). However, deriving robust historical population trends is challenging when scientific monitoring data are scarce or unavailable (Pauly, 1995; Sáenz-Arroyo et al., 2005; Beaudreau & Levin, 2014). This is further aggravated in data-poor contexts, when a species is impacted by illegal, unreported, or unregulated exploitation. Common data-poor contexts include small-scale fisheries and subsistence hunting (Moller et al., 2004; Duffy et al., 2016; Selgrath, Gergel & Vincent, 2018). This challenging situation has led to increased interest in Local Ecological Knowledge (LEK), including traditional knowledge (TK) of indigenous peoples, to better understand long-term environmental change and human-environment interactions (Johannes, 1981; De Castro et al., 2014; Bao & Drew, 2017; Lee et al., 2018; Barrios-Garrido et al., 2018).

LEK can be defined as place-based empirical knowledge, held by a specific group of people about their surrounding environments and biota (Bélisle et al., 2018). LEK does not require that knowledge-holders be indigenous, nor embedded in a broader shared culture, and thus can be applied to people and communities with relatively short histories of interactions with a specific environment (cf. Narchi et al., 2014). LEK data have been used in combination with official records and historical documentation to reconstruct long-term abundance trends of exploited marine species in multiple contexts (Jackson et al., 2001; Sáenz-Arroyo et al., 2005; Beaudreau & Levin, 2014; Lee et al., 2018). LEK also provides baseline data that fill knowledge gaps which cannot be addressed through natural sciences alone (Mukherjee et al., 2018; Mason et al., 2019). Examples include knowledge of ecological change over broad time-scales (Sáenz-Arroyo et al., 2005; Lee et al., 2018), traditional and local resource use (Johannes, 1981; Barrios-Garrido et al., 2018), and conceptual frameworks for ecological modeling (Ainsworth, 2011; Bélisle et al., 2018). However, clear methodological guidelines, based on robust methods from social and natural sciences, are needed to reliably integrate LEK with scientific ecological data in

conservation science ([Mukherjee et al., 2018](#); [Young et al., 2018](#); [Moon et al., 2019](#)). This includes developing approaches to collate and validate information from diverse knowledge sources, and forming interdisciplinary teams with expertise appropriate for the methods being used ([St. John et al., 2014](#); [Sutherland et al., 2018](#)).

We present a case study of the East Pacific green turtle (*Chelonia mydas*, hereafter green turtle) in Bahía de los Ángeles (BLA), Baja California, Mexico, to demonstrate a novel framework that can be adapted to long-lived, exploited taxa to evaluate abundance trends in data-poor scenarios. We used ethnography to document LEK, and developed an ad hoc epistemological approach to synthesize and quantify LEK data using generalized linear models (GLMs) and nonlinear least squares regression (NLR) to reconstruct long-term *C. mydas* abundance. Our model established baseline abundance, described historical declines, and evaluated how human impacts contributed to current species population status.

The complexity of the green turtle's life history makes it particularly challenging to evaluate its conservation status. Generation times are up to 50 years, they are highly migratory, and life stages occupy multiple habitats separated by hundreds or thousands of kilometers, often in different countries. Globally, abundance data are skewed towards nesting beaches, which only quantify nesting females ([Seminoff & Shanker, 2008](#); [Godley et al., 2010](#)). For the Eastern Pacific population, nesting data have been collected since 1980 at the primary nesting beach in Colola, Michoacán, Mexico (~1,500 km from BLA) ([Delgado-Trejo, 2016](#)). However, there are substantial knowledge gaps for foraging habitats, which are critical for several reasons. Foraging habitats include pre-reproductive life stages—which are the most abundant life stages in the population—along with adults of both sexes. Furthermore, foraging habitats are where green turtles spend the majority of their life: juveniles may spend 20 years or more in foraging grounds until reaching maturity, and adults reside at feeding grounds during non-breeding periods ([Seminoff, Resendiz & Nichols, 2002](#); [Seminoff & Shanker, 2008](#); [Senko et al., 2019](#)). Thus, expanding data on foraging habitats is of utmost importance for a holistic understanding of population status ([Chaloupka et al., 2008](#); [Mazaris et al., 2017](#); [Wildermann et al., 2018](#)).

Green turtles are listed as Endangered by the IUCN and Mexican law as a result of population collapse due to a largely unregulated fishery between the 1950s and 1980s ([Diario Oficial de la Federación, 1990](#); [IUCN, 2019](#); [SEMARNAT, 2010](#)). Populations in the Eastern Pacific have increased since the early 2000s thanks to decades of nesting beach protection at Colola starting in the late 1970s, coupled with expanded efforts to limit egg harvests, direct captures, and poaching throughout the species' range in Mexico ([Delgado-Trejo, 2016](#); [Seminoff et al., 2015](#)). These efforts were fortified by the 1990 ban on all sea turtle use in Mexico, which established a legal framework to prevent harvests ([Diario Oficial de la Federación, 1990](#); [SEMARNAT, 2010](#)). However, abundance data and long-term trends prior to population collapse are needed to contextualize current population levels ([Early-Capistrán et al., 2018](#); [Seminoff et al., 2008](#)).

Starting with an overarching research question (e.g., What was the baseline green turtle abundance, and how did it change over time, before scientific monitoring?), we carried out

background research with natural and social science perspectives to gain a broad understanding of the research topic (Crandall *et al.*, 2018). We then used an iterative approach to document LEK through ethnography, and synthesized LEK-data for integration with ecological modeling to provide a consistent long-term time-series of green turtle abundance data that can inform conservation.

## METHODS

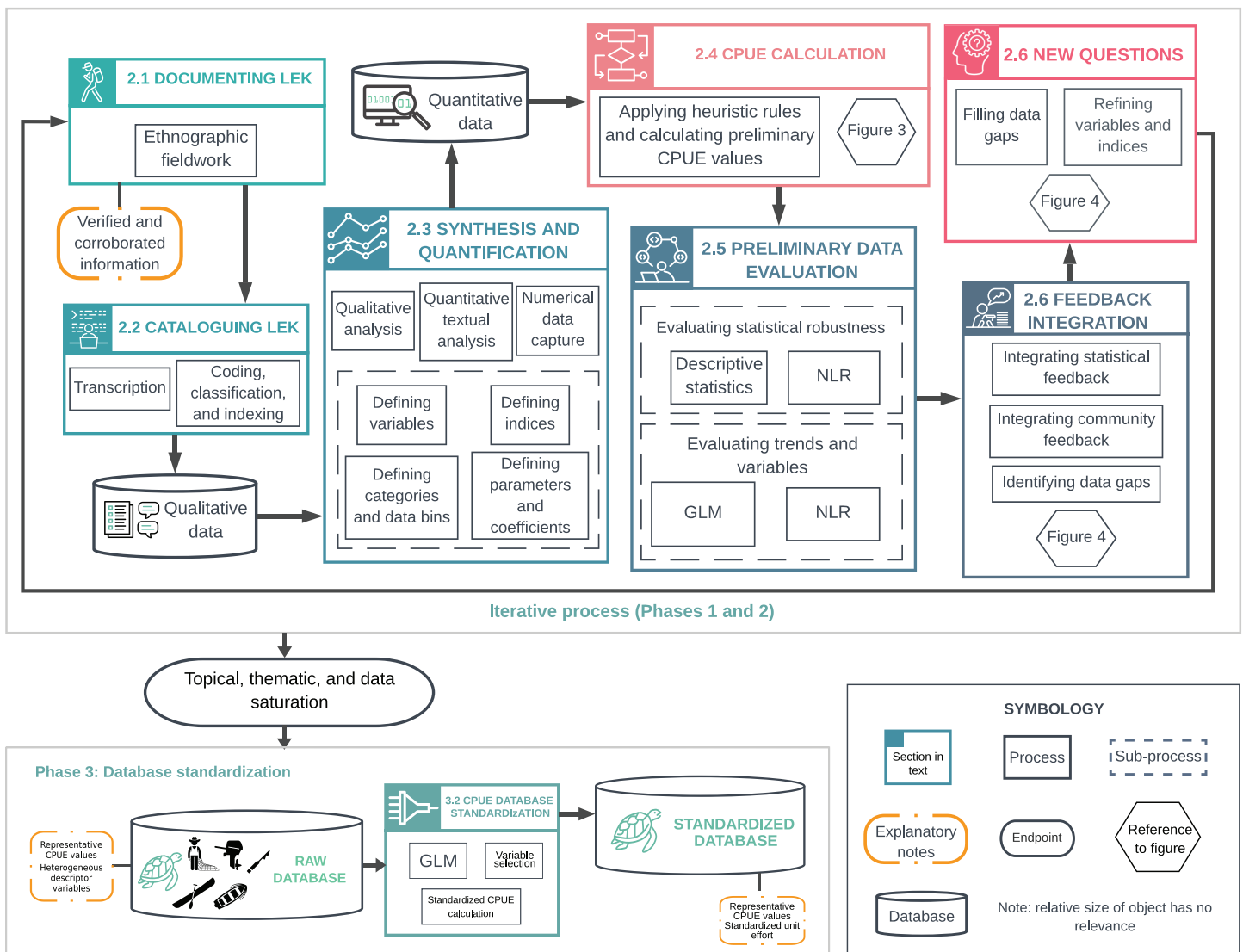
We present a flexible approach for generating green turtle abundance estimates from LEK that can be modified for long-lived species with a history of human use. Our approach consists of four phases: (1) background research and experimental design; (2) an iterative process of LEK documentation, synthesis, and quantification; (3) database standardization and validation; and (4) statistical analysis and modeling of the standardized database (Fig. 1). Interdisciplinary teams can ensure that quality and reliability standards are met across fields (Tengö *et al.*, 2014; St. John *et al.*, 2014; Sutherland *et al.*, 2018). Detailed accounts of methods and tools are available in Supporting Information (henceforth, SI) (Article S1).

### Phase 1: background research and experimental design

#### *Study site*

To demonstrate our methods, we used the case of the green turtle in Bahía de los Ángeles (BLA), Baja California, Mexico (28°57'6.90"N, 113°33'44.76"W), an index foraging area in the Gulf of California (Seminoff *et al.*, 2003, 2008). We define an index foraging area as a site that (i) has aggregations of turtles in the marine environment that represent a significant proportion of the regional population, and (ii) has been monitored systematically and constantly over a prolonged period of time (>10 years). In-water scientific monitoring in this foraging area began in 1995, after population collapse (Seminoff *et al.*, 2003, 2008). Contemporary scientific monitoring uses catch-per-unit-effort (CPUE) as a measure of abundance (Seminoff *et al.*, 2008).

Green turtles have been a key food source for humans in the arid Baja California peninsula since the earliest phases of human occupation at least 12,000 years ago (cf. Early-Capistrán, 2014). From the late 18th century until the early 1950s, green turtle harvests were primarily subsistence-oriented. Turtles were harpooned from small, wooden canoes propelled with oars or paddles. During the 1960s, the economic and demographic growth along the U.S.-Mexico border led to an increased market for green turtle meat in Mexican border cities. BLA was a key supplier within this trade, and was able to meet demands as the introduction of outboard motors, fiberglass vessels, and set-nets increased cargo volume and catch efficiency. Additionally, improvement of transport and communication infrastructure facilitated market access (Early-Capistrán *et al.*, 2018). The fishery collapsed in the 1970s, green turtle licenses were suspended in 1983 as populations reached dangerously low levels, and all sea turtle fishing in Mexico was banned in 1990 (Márquez, 1996; Seminoff *et al.*, 2008).



**Figure 1** Overview of methodological processes used to document, synthesize and quantify Local Ecological Knowledge (LEK). The upper box illustrates the iterative process described in Phases 1 and 2. This includes LEK documentation and synthesis; analysis with descriptive statistics, generalized linear models (GLMs), and nonlinear regression (NLR); and integration of feedback from statistical analyses and local collaborators. This iterative process was repeated until reaching topical, thematic, and data saturation, and until model fitting did not provide significant new information. The lower box illustrates catch-per-unit-effort (CPUE) data standardization (Phase 3). The raw database (Raw CPUE Database Analysis) contained average, representative CPUE values for a given year, and heterogeneous descriptor variables. We standardized CPUE values using GLMs (CPUE Database Standardization) to (i) remove most of the variation not attributable to changes in abundance, and (ii) generate CPUE values that could be compared over time.

Full-size DOI: [10.7717/peerj.9494/fig-1](https://doi.org/10.7717/peerj.9494/fig-1)

### Background research

This research is part of an on-going collaborative process in the community of BLA which began in 2012 and has included ethnographic and historiographical research related to human–ocean interaction, along with a review of scientific literature (*Early-Capistrán et al., 2018*). Background research helped define specific research questions, identify challenges in the study design and methods, and develop a general approach for



integrating multiple forms of knowledge ([Crandall et al., 2018](#); [Early-Capistrán et al., 2018](#)).

Historiographical research situates biological questions in a socio-historical context, providing information on a species' past abundance which can be correlated with time-frames, social processes, and management regimes ([Article S1](#)) ([Crandall et al., 2018](#); [Sáenz-Arroyo et al., 2005](#)). Historiographical research helped us understand human-green turtle interactions in BLA over the past three centuries, identify the early 1960s as a period when human impacts precipitated a major decline in green turtle abundance in BLA, and establish the early 1950s as a time-frame for reconstructing baseline abundance before large-scale commercial exploitation ([Early-Capistrán et al., 2018](#)).

Long-term collaboration with the community of BLA was fundamental for previously establishing the rapport and working trust necessary to conduct transdisciplinary research. Long-term engagement has also helped us acquire sensitivity to the cultural context, gain an understanding of social conditions, and gather locally-relevant information to define research questions and design ([Bernard, 2011](#); [Crandall et al., 2018](#)). We also established a network of local collaborators, whom we define as knowledgeable community members willing to share their knowledge and expertise ([Crandall et al., 2018](#)). Due to the fact that ecological knowledge is differentially acquired by social actors, we constructed a heterogeneous network of social actors with diverse types of knowledge that, when nested together, construct the ecological knowledge around green turtle abundance (cf. [Brown, 2010](#)).

### **Experimental design**

#### *Qualitative methodology*

Ethnography was our primary data-gathering methodology. This holistic approach to the study of social systems uses a varied toolkit to generate both qualitative and quantitative data ([Table 1](#); [Article S1](#); [Table S1](#)). Ethnography requires rapport, sensitivity to the cultural context, and developing an understanding of the social system on its own terms. Data are gathered broadly over topic areas and new questions are developed continuously ([Bernard, 2011](#); [Early-Capistrán et al., 2018](#)). Ethnography also helps identify biases by analyzing data within a social and historical context ([Drury, Homewood & Randall, 2011](#)). Ethnographic data are systematized, cross-referenced, verified, and subject to analysis and meta-analysis ([Bernard, 2011](#)).

We chose ethnography because (i) the high degree of socio-environmental complexity required detailed information on diverse topics; (ii) sea turtle fishing is currently illegal in Mexico and its inquiry requires working trust, long-term engagement, and confidentiality; and (iii) ethnography provides more detailed and reliable information on sensitive issues than is provided by questionnaires ([Drury, Homewood & Randall, 2011](#); [St. John et al., 2014](#)). Research was designed in compliance with the ethical guidelines of the International Society of Ethnobiology ([Articles S1 and S2](#)) ([International Society of Ethnobiology, 2006](#)) and approved by the Bioethics Committee of the Centro de Investigación Científica y de Educación Superior de Ensenada (Approval Number 2S.3.1).



**Table 1** Methods used for data collection during ethnographic field work. Sources: *Bernard (2011)*; *Crandall et al. (2018)*; *Early-Capistrán et al. (2018)*.

Method	Definition	Example of applications	Practical implications
Participant observation	Studying a social group through a combination of direct observation and immersion in group activities as an active participant	Participating in and documenting sport-fishing trips led by former green turtle fishers	All observations are compiled in field notes and journals, including, but not limited to research topics
Informal interviews	Interviews without structure or control, often conversations held during the course of fieldwork	Conversations with fishers or their family members recorded in written notes	Recorded in field notes and field journals
Semi-structured interviews	Interview based on a flexible list of written questions or topics that need to be covered. The interviewer maintains discretion to follow new leads	Contributors were interviewed using an interview guide with recurring topics focused on the green turtle fishery	Recorded in audio or video with the collaborators' consent
In-depth interviews	Aimed at obtaining detailed understanding of the topic of interest. Participants can communicate more freely and provide more detailed descriptions than with semi-structured interviews	Experts and key local collaborators were interviewed in-depth on specific topics related to green turtle fishing or abundance (e.g.: fishing gear, green turtle commerce, etc.)	Recorded in audio or video with the collaborators' consent
Focus groups	Moderated discussions with small groups (<10 people) on a particular topic	Focus group discussions with members of a fishing crew to discuss how green turtle abundance changed over the course of their careers	Recorded in audio or video with the collaborators' consent
Oral histories	In-depth interviews about life stories, experiences, and eyewitness accounts	Interviewing experts on their life history and their experience as green turtle fishers	Recorded in audio or video with the collaborators' consent
Participatory mapping	Contributors draw maps, locate key places on maps, or locate key sites together with researchers	Visiting key green turtle fishing spots and recording coordinates with GPS	Recorded in notes, digital maps, GIS or printed maps
Social network analysis	Identifying the structure of social relations	Documenting kinship and work relations among green turtle fishers and merchants	Recorded in notes and graphs
Discourse analysis	Analysis of communicative content and structure focused on how meaning is constructed and how power functions in a society	Analyzing discourse on regulation or conservation to identify biases that could affect how fishers report on turtle catches	Analysis of ethnographic materials; feedback integrated into new questions

We defined three social groups within the community and documented their knowledge. Fishers who participated in the legal green turtle fishery before 1990 (henceforth, turtle fishers) constituted the target population and provided the majority of specialized LEK related to human-turtle interaction. This group was the main focus of ethnographic research and contributed the majority of the qualitative and numerical data. Key local collaborators—defined as community members with expertise in particular topics—provided important complementary and contextual information on topics such as local history, commerce, or foodways, among others. Finally, we gathered additional complementary data from members of the community at large (henceforth, community members), including fishers' families, green turtle merchants, local authorities, commercial and sport fishers, and conservation workers, to understand and incorporate multiple perspectives. Methods and sample sizes used for each of these groups are discussed in detail in “Documenting LEK”.

**Box 1 Primary topic areas in interview guides.**

1. Biographical data and career history
  - Year of birth
  - Years in the community
  - Years as a fisher
  - Years in the green turtle fishery
  - Crew members and fishing merchants with whom they worked
2. Sea turtle consumption and commerce
  - Domestic sea turtle consumption dynamics (before 1990 ban)
  - Market dynamics for sea turtle sale (how, where, and how often turtles were shipped)
  - Commercial dynamics (how turtles were sold, prices, working relationships, etc.)
3. Sea turtle catches and sizes
  - Maximum and minimum catches
  - Frequency of aggregations and large catches
  - Average catches
  - Perceived changes in abundance
  - Size distribution (maximum and mode sizes, frequency of catching large turtles)
  - Sea turtle ethnobiology (effects of seasonality, tides, green turtle behavior, etc.)
4. Spatial distribution of fishing
  - Frequently used fishing grounds
  - Hot-spot and aggregation dynamics
  - Changes in use of fishing grounds across time
  - Distances and travel times to fishing grounds
5. Fishing effort and technology
  - Use and efficiency of different gear types/gear designs
  - Use of different vessels
  - Use of different propulsion systems

We designed flexible interview guides for use in semi-structured and in-depth interviews based on previous ethnographic research on sea turtle use in BLA ([Early-Capistrán et al., 2018](#)). Interviewers M.M.E.C. and G.G.M. used these guides as a roadmap for the interviews, allowing respondents to be thorough and make associations between questions, and to include new topics and questions according to interview progress (cf. [De Castro et al., 2014](#)). Interview guides covered five main topic areas: (1) biographical profile and career history; (2) sea turtle consumption and commerce; (3) trends in sea turtle captures and sizes; (4) spatial distribution of sea turtle fishing; and (5) fishing effort and technology ([Box 1](#)). To prompt recollection of dates, questions were associated with important events in local collaborators' lives ([Article S1](#)). We piloted questions with local fishers outside the target population ( $n_{\text{pilot}} = 2$ ) and constantly refined

the questions to ensure that they were locally contextualized and elicited meaningful answers (Bernard, 2011; Drury, Homewood & Randall, 2011; Young et al., 2018).

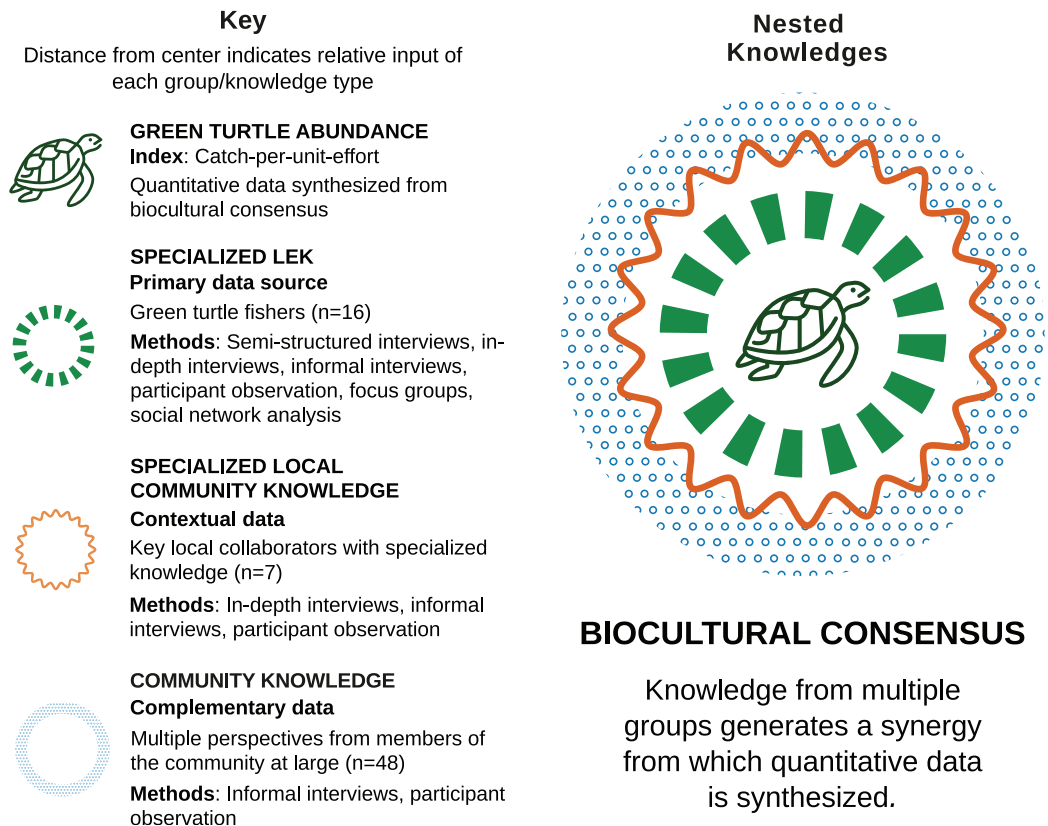
#### *Reconstructing green turtle abundance through collective knowledge*

Defining an approach to estimate green turtle abundance based on CPUE was a key challenge. Although CPUE is a crude measure of changes in exploited populations (López-Castro et al., 2010), we used it because (i) it is the only available metric of current abundance and (ii) CPUE is an accepted proxy for abundance for IUCN Red Listing (IUCN, 2019; O'Donnell, Pajaro & Vincent, 2010).

Adequate assessment of CPUE as a measure of abundance requires detailed understanding of the fishery and the variables that affected it (Moller et al., 2004). The skilled turtle fishers of BLA almost always targeted high-density locations (hot-spots) and aggregations, and thus maximized CPUE by optimizing fishing patterns based on empirical knowledge of environmental conditions and green turtle behavior (Early-Capistrán et al., 2018). Consequently, turtle fishers' expertise allowed for high CPUE events over time despite declining overall abundance (hyper-stability), underscoring the need to (i) account for this non-random search behavior and (ii) understand central CPUE trends rather than exceptional catches (Article S1; Fig. S1) (Anticamara et al., 2011; Early-Capistrán, 2014; Maunder & Punt, 2004; Selgrath et al., 2018; Walters, 2003).

This scenario is challenging, as (i) interviewees' memory of "typical" events may be less accurate than that of salient events and (ii) high variability in CPUE and changes in fishing efficiency can mask overall abundance trends (Maunder & Punt, 2004; De Damasio et al., 2015; Sáenz-Arroyo & Revollo-Fernández, 2016). Thus, we designed our methodology to calculate CPUE based on multiple sources rather than individual recollections. We also aimed to identify and account for sources of variation in CPUE that could bias proportionality with abundance, and to construct adequate proxies for variables such as spatial distribution of fishing, differences in gear types, and changes in fleet conditions (Walters, 2003; Maunder & Punt, 2004; Anticamara et al., 2011; Selgrath, Gergel & Vincent, 2018).

We approached CPUE as a component of a holistic dataset on human-environment interaction, and aimed to synthesize quantitative values on the basis of biocultural consensus, which we define as the pooling of information for evaluating shared environmental perceptions constructed by the summation of individual, community, specialist, and holistic types of knowledge. Biocultural consensus is a synergistic, interconnected set of contents and types of knowledge (c.f. Brown, 2010) in which the resulting knowledge is greater than sum of its parts. In this case, we used knowledge from all three social groups (turtle fishers, key local collaborators, and community members) as inputs for constructing biocultural consensus. Our ethnographic research was primarily focused on turtle fishers, who provided the majority of qualitative and numerical data, as well as specialized LEK related to human-turtle interaction. Key local collaborators and community members provided contextual and complementary data (Fig. 2). Biocultural consensus helped build conceptual frameworks for modeling, establish



**Figure 2** Biocultural consensus as a synergy of interconnected knowledge (adapted from *Brown, 2010*). Biocultural consensus was constructed with multiple and complementary contents and knowledge types from three different social groups. Sample sizes, ethnographic methods, and interview methods used with each group are provided. The target population of turtle fishers were the group with which we collaborated most intensively and which provided the majority of LEK, as well as qualitative and numerical data. Turtle fishers' knowledge was complemented with knowledge from of key local collaborators and community members. Biocultural consensus helped build conceptual frameworks for modeling, establish limits and assumptions, estimate model parameters, and validate model outputs.

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limits and assumptions, estimate model parameters, and validate model outputs (*Bélisle et al., 2018*).

As the primary response variable, we aimed to calculate representative values of CPUE during a specific year with the initial definition:

$$\text{CPUE} = \text{number of turtles caught/unit effort} \quad (1)$$

For initial inquiry, we used the working definition of one unit effort as one night of fishing (~12 h) with either a harpoon or a set-net (*Maunder & Punt, 2004*). We continually refined and updated this definition as we gained further information on fishing technology, effort, and efficiency through the iterative feedback process between qualitative data, NLR, and GLMs (Phase 2). We then standardized CPUE estimates to account for differences in gears and changes in efficiency (Phase 3). As the final result of the iterative feedback process, we obtained standardized, representative mean CPUE values for a specific year, based on biocultural consensus of green turtle captures.

**Table 2 Tools and criteria for the model fitting and selection processes.** Throughout the iterative process, we used nonlinear regression to describe catch-per-unit-effort trends over time, and generalized linear models to identify significant predictor variables. Residual analyses were used to ensure that model assumptions were met, and to evaluate goodness of fit and robustness.

Process	Software	Model selection criteria	Residual analyses
Preliminary model selection and starting values	LABFit 7.2.49	$R^2$ value	–
Nonlinear regression (NLR)	R 3.4 ( <i>nlstools</i> and <i>easynls</i> package)	$R^2$ value Robust residuals: $e_i \sim N(0, \sigma^2)$	Normality: Shapiro–Wilk test, $p > 0.05$ Mean = 0: $t$ -test, $p > 0.05$ Homogeneity of variance: Levene’s test, $p > 0.05$ Randomness: runs test, $p > 0.05$ Auto-correlation: Pearson correlation test (residuals vs. lagged residuals), $p > 0.05$ (i.e., $H_0: \rho = 0, H_a: \rho \neq 0$ )
Generalized linear model (GLM)	R 3.4 ( <i>nlme</i> , <i>lmtest</i> and <i>car</i> packages)	Significant predictor variables ( $p < 0.05$ ) $D^2$ value Low relative AIC Robust residuals: $e_i \sim N(0, \sigma^2)$	Normality: Shapiro–Wilk test, $p > 0.05$ Mean = 0: $t$ -test, $p > 0.05$ Homogeneity of variance: Levene’s test, $p > 0.05$ Randomness: runs test, $p > 0.05$ Auto-correlation: Durbin–Watson test, $p > 0.05$

### Quantitative methods

Throughout the iterative process, we used descriptive statistics for exploratory data analysis and to identify outliers (Zar, 2014). We used NLR to describe CPUE trends over time and GLMs to identify significant predictor variables. We also integrated residual analysis to ensure that model assumptions were met and to evaluate goodness of fit and robustness. We ensured that residuals met the assumptions of zero mean, normal distribution, homoscedasticity, and independence ( $e_i \sim N(0, \sigma^2)$ ) (Table 2; Article S1) (Maunder & Punt, 2004; Ritz & Streibig, 2008). In response to residual auto-correlation found during preliminary phases and exploratory data analysis, as is common with time series data, we integrated residual correlation structures to GLMs with residual auto-correlation during the final stages of standardization (Zuur, 2009). All models reported in the Results meet the assumptions for robust residuals according to the criteria described in Table 2 (see also Article S1).

It should be noted that the statistical treatment is applied to the data series synthesized from biocultural consensus. We used all available information to amass a year by year remembrance of turtle captures by combining fishers’ knowledge with that of key local collaborators and community members. Thus, our synthesized data is not derived directly from the individual, yearly recollections of specific fishers, but instead are the result of collectively generated and corroborated knowledge. Likewise, statistical analyses were not conducted in relation to the social groups themselves (aside from simple demographic description), but rather to the quantified data synthesized from their collective knowledge, which included sea turtle captures as well as descriptor variables, coefficients, and indices (Table 3).

## Phase 2: recording, synthesizing, and quantifying LEK

### Documenting LEK

M.M.E.C. and G.G.M. compiled ethnographic data in BLA over three field seasons (spring 2017, summer 2017, and spring 2018) and 57 working days. We obtained oral

informed consent from all participants prior to the start of interviews, and recorded interviews in audio or video and took technical photographs when possible ([Article S1](#); [Tables S2](#) and [S3](#)) (*International Society of Ethnobiology, 2006*). We chose oral consent as it was not deemed culturally appropriate to ask participants to sign a consent document and because some participants were not comfortable with written language (*International Society of Ethnobiology, 2006; Wedemeyer-Strombel et al., 2019*). We conducted all interviews in Spanish—our primary language and that of the collaborators—and transcribed recorded interviews in digital format (.txt). We also compiled field journals in digital format (.txt), recording all observations in detail ([Article S1](#)).

We validated ethnographic data through triangulation among (i) participants (e.g., data were independently corroborated and verified by multiple local collaborators), (ii) sources (e.g., documents, photographs, scientific literature, etc.), and/or (iii) methods (e.g., interviews, archive research, etc.). Once processed, we member-checked data for reliability by asking local collaborators from all groups if our themes or categories were locally relevant and congruent. We also asked local collaborators to identify data gaps, and inquired if overall accounts and processes were described in a manner that was realistic and accurate (*Creswell & Miller, 2000; Tengö et al., 2014*). Prolonged engagement in the field allowed us to compare interview data with observations, and helped build trust so that participants were comfortable disclosing information, increasing reliability in responses (*Bernard, 2011*).

We identified turtle fishers using a deliberate hierarchical sampling method (*Bernard, 2011*), Turtle fishers are a small group of the oldest fishers in the community, between 55 and 85 years of age ( $N_{\text{fishers}} = 17$ ). We interviewed 94% of turtle fishers, as one fisher chose not to participate. All fishers in the population and sample were men. With this target group, we continuously carried out participant observation, and conducted 17 semi-structured interviews (at least one per person), along with 27 informal interviews. Within this target population, we identified a subset of seven expert LEK holders, which we defined as turtle fishers recognized as experts by at least two peers, and whose empirical and specialized knowledge can be used as a basis for inferences and assessments about their surrounding environments and biota (cf. *Bélisle et al., 2018*). With the group of expert LEK holders, along with the aforementioned methods, we conducted seven in-depth interviews and one focus group discussion to gather specialized data (*Tengö et al., 2014*).

We identified key local collaborators ( $n_{\text{kdc}} = 7$ ) through purposive and respondent-driven sampling (*Bernard, 2011*). Key local collaborators were primarily older (>63: 71%) and included women (43%) and men (57%). We continuously carried out participant observation with this group, and conducted four in-depth interviews and 23 informal interviews. Topics included: local history, economy, commerce, and foodways; marine and terrestrial ethnobiology and conservation; and commercial and sport fishing, among others, which provided valuable information for situating green turtle fishing within a broader socio-ecological context (*Crandall et al., 2018*).

We selected local collaborators from the community at large ( $n_{\text{cm}} = 48$ ) through a combination of cluster sampling and self-selection (*Bernard, 2011*). They represented ~8%



**Table 3** Variables, coefficients, and indices.

Variable or coefficient	Type	Index	Source
Year of birth	Numerical	Date	Standard question in interviews
Dates working in the green turtle fishery	Range	Interval of dates	Standard question in interviews
Experience in the green turtle fishery	Ordinal	1 = 1–5 years 2 = 6–10 years 3 = 11–15 years	Binned from dates working in the fishery
Generation	Categorical	1 = Fishers who worked in commercial development and commercial fishing stages 2 = Fishers who worked during the collapse stage 3 = Fishers who worked through all stages	Category of cohorts of fishers defined based on the fishery stages in which the contributor worked
Fishery stage	Categorical	1 = Commercial development 2 = Commercial fishing (harpoon) 3 = Commercial fishing (nets) 4 = Collapse	Defined based on qualitative data on the fishery
Year	Numerical	Date for which the average CPUE is being described	Obtained directly from interviews (numerical value) or calculated based on heuristic rules (details in S.I.)
Fishing gear	Ordinal	1 = Harpoon 2 = Short set-net (~100 m) 3 = Long set-net (~200 m)	Binned from interviews or inferred based on heuristic rules
Harpooner skill coefficient	Percentage	Percentage of success (50–99%) <sup>a</sup>	Obtained from interview data and assigned to contributors based on social network analysis
Number of nets	Numerical	Number of nets used <sup>b</sup>	Obtained directly from interviews or inferred based on heuristic rules
Vessel type	Ordinal	Type of vessel used 1 = Wooden canoe (12–15 ft length) 2 = Fiberglass skiff (20–22 ft length) 3 = Boat (variable length)	Binned from interviews or inferred based on heuristic rules
Vessel capacity	Ordinal	Gross vessel tonnage 1 = Less than 1 ton 2 = 1–1.5 tons 3 = Greater than 1.5 tons	Binned from interviews or inferred based on heuristic rules
Propulsion <sup>c</sup>	Categorical	1 = Oars 2 = Motor (5–10 horse-power) 3 = Motor (15–40 horse-power)	Obtained directly from interviews or inferred based on heuristic rules
Trip duration <sup>c</sup>	Numerical or interval	Number of days between leaving port and returning with a catch of turtles at vessel capacity Minimum limit: 1 day Maximum limit: 10 days	Obtained directly from interviews or inferred based on heuristic rules (S.I., Eqn. S1, S2)
Fishing time	Numerical	Number of nights spent fishing on a trip of regular duration	Obtained directly from interviews or inferred based on heuristic rules (S.I., Eqn. S1, S2)
Average CPUE	Numerical	Average number of turtles caught in one night during a specific year	Obtained directly from interviews (numerical value) or calculated based on heuristic rules

**Notes:**<sup>a</sup> Not assigned to captures with nets.<sup>b</sup> Not assigned to harpoon captures.<sup>c</sup> Proxies for spatial distribution of fishing.

of the population of BLA and included women (42%) and men (58%). Ages ranged from 18 to 93, with young (18–39: 35%), middle-aged (40–62: 37%), and older (>63: 28%) participants. While we did not inquire about income given local social taboos, local collaborators came from across all class strata with schooling varying from individuals without formal schooling to graduate degree holders. The group included both long-term residents (89%) and short-term residents (11%) such as conservation workers and government employees. This diverse group provided a broad view of perspectives and topics to complement and contextualize information from the target population of turtle fishers. With this group, we continuously carried out participant observation and conducted 72 informal interviews.

### ***Cataloguing LEK***

We processed and coded all field journals and interview transcriptions following a standardized protocol. We used footnotes to separate observations from analysis, and for cross-referencing. Cryptic indicators ensured local collaborators' anonymity ([Bernard, 2011](#)). We used cultural material codes ([Murdock et al., 2008](#)) to categorize ethnographic data, with customized codes for topics and themes specific to this research. We indexed text entries using hashtags (#) to mark relevant topics (e.g., #fishing\_gear), including ordinal codes (e.g., #max\_cpue; #min\_cpue) to classify information for data-binning ([Article S1](#); see [Table S4](#) for an example field journal entry). Along with data compiled in the 2017 and 2018 field seasons, we coded and indexed ethnographic materials collected since 2012 for integration into the qualitative database ([Article S1](#); [Tables S2](#) and [S3](#)). Coding allowed us to break down qualitative data into analytical variables and raw values ([Strauss & Corbin, 1994](#)). Digital files allowed for analyzing large volumes of information by facilitating topic-specific searches, generating a corroborated, systematized, and cross-referenced qualitative database ([Bernard, 2011](#)).

### ***Synthesizing and quantifying LEK***

#### *Qualitative analyses*

We used qualitative textual analysis and discourse analysis to decipher the cultural, historical, and political dimensions of the research topic; to identify potential sources of bias; and to understand categories, processes, and connections ([Crandall et al., 2018](#)) ([Table S1](#)). We captured raw numerical data from interviews ([Article S1](#); [Table S4](#)), and used Quantitative Textual Analysis tools in R 3.4 (*wordcloud*, *tm*, and *SnowBallC* packages) to identify themes and patterns over large volumes of text, for a general overview ([Bernard, 2011](#); [R Core Team, 2019](#)) ([Article S1](#); [Figs. S2](#) and [S3](#)). These themes helped us to identify potential descriptor variables, indices, and topics for inquiry.

#### *Quantifying LEK data*

We defined explanatory variables for CPUE based on qualitative data ([Table 3](#)). We generated initial indices for each variable based on the degree of detail and variation observed in interview responses, and defined standardization and binning procedures ([Fig. 1](#)).



We established four stages for the BLA green turtle fishery based on fishery landing statistics and previous research (*Early-Capistrán et al., 2018; Selgrath, Gergel & Vincent, 2018*): (1) commercial development; (2) commercial fishing (harpoons); (3) commercial fishing (nets); and (4) collapse (*Table 4*). Qualitative data allowed for inferring that (i) fishing technology across the fleet was similar within each stage; (ii) at all stages, fishers would make trips of varying duration until reaching vessel capacity or exhausting food and water supplies; and, thus, (iii) CPUE could be calculated based on the knowledge of fisheries stages, trip duration, fishing gear type, displacement time, and vessel capacity (*Article S1*). This framework allowed us to (i) bin data and standardize variations in expertise and response terms, (ii) systematically complement the knowledge of less experienced fishers with that of expert LEK holders, and (iii) account for changes in fishing technology, effort, and efficiency over time (cf. *Maunder & Punt, 2004*).

We generated digital (.txt) files to summarize categorical, ordinal, and numerical data for each fisher (*Article S1; Table S5*). Using social network analysis (*Bernard, 2011*), we situated each fisher in relation to their fishing crew and extended family (*Table 1*). Ethnographic and LEK data provided us with numerical anchor values and limits for variables during each stage (*Bélisle et al., 2018*) (*Article S1*).

### ***CPUE calculation and preliminary database generation***

To deal with variability, we used heuristic rules to make systematic inferences based on the knowledge of expert LEK holders (*Fig. 3*). This framework allowed us to calculate a central tendency based on collectively-generated knowledge and biocultural consensus rather than individual recollection, thus reducing individual cognitive bias (*Article S1*).

We converted captures reported by weight to number of turtles by dividing vessel capacity by mode of turtle mass (50 kg) reported by fishers and corroborated with monitoring data (*Early-Capistrán et al., 2018*) (*Article S1*). While turtle size was highly variable and likely declined in response to increasing fishing effort (*Table 4*), mixed juvenile/adult foraging groups with a slight juvenile bias—such as BLA, where ~56% of individuals are juveniles (*Seminoff et al., 2003*)—are present in green turtle foraging habitats worldwide (*Seminoff et al., 2015*). Thus, we consider our assumption regarding size distribution to be adequate given the nature of the data (*Table 4; Article S1*).

### ***Preliminary data evaluation***

We estimated CPUE and descriptor variables through an iterative process. We stored data in .csv format and carried out all analyses in R 3.4 unless otherwise specified (*R Core Team, 2019*). We analyzed descriptive statistics to evaluate statistical robustness by checking data distribution, evaluating normality (Shapiro–Wilk  $p > 0.05$ ), and identifying outliers ( $\pm 2$  SD) (*Zar, 2014*). Each CPUE data point was linked to a summary of qualitative and numerical data for a specific collaborator, and outlying data were contextualized and evaluated (*Article S1; Table S5*). Over the course of the iterative process, we discarded three CPUE values from fishers who (i) had less than 1 year of experience and (ii) were very

**Table 4** Fishery stages and characteristics.

	Commercial development (1950–1959)	Commercial fishing (harpoons) (1960–1965)	Commercial fishing (nets) (1966–1972)	Collapse (1974–1982)
General characteristics	First years of the commercial fishery, with limited technology and fishing effort	Intense growth in demand leads to declining captures	Increasing fishing effort and efficiency, declining captures	Commercial collapse. Species abundance near extinction.
Regulation	Unregulated	Unregulated	Limited regulation: minimum size, permit restrictions, seasonal bans Temporary ban (1971)	Highly regulated: minimum size, permit restrictions, seasonal bans, nesting beach protection (1980-present) Green turtle licenses suspended (1983)
Gear type	Harpoons	Harpoons	Set-nets	Set-nets
Fleet conditions	Wooden canoes Oars or paddles	Wooden canoes 5–10 horse-power outboard motors	Canoes or skiffs 5–10 horse-power outboard motors	Fiberglass skiffs 15–45 horse-power outboard motors
Spatial distribution of fishing <sup>a</sup>	Overnight trips close to port are frequent	Motors allow faster displacement to farther fishing grounds Occasional trips >50 nautical miles	Trips >50 nautical miles are frequent Expeditions >100 nautical miles are frequent (canoes or skiffs off-loading to boats)	Trips >50 nautical miles are frequent
Size distribution <sup>b</sup>	Turtles ~150 kg caught frequently (spans of weeks/months) Mode weight: 50 kg	Turtles ~150 kg caught frequently (spans of weeks/months) Mode weight: 50 kg	Turtles 100–150 kg caught occasionally (spans of seasons/years) Mode weight: 50 kg	Turtles 100–150 kg caught rarely (spans of years) Mode weight: 50 kg
Fishing efficiency	Low	Low/Moderate	Moderate	High
Fishing effort	Low	High	High	Low
Commercial demand	Moderate	High	High/moderate	Moderate
Profitability	High	High	High/Declining	Not profitable

**Notes:**

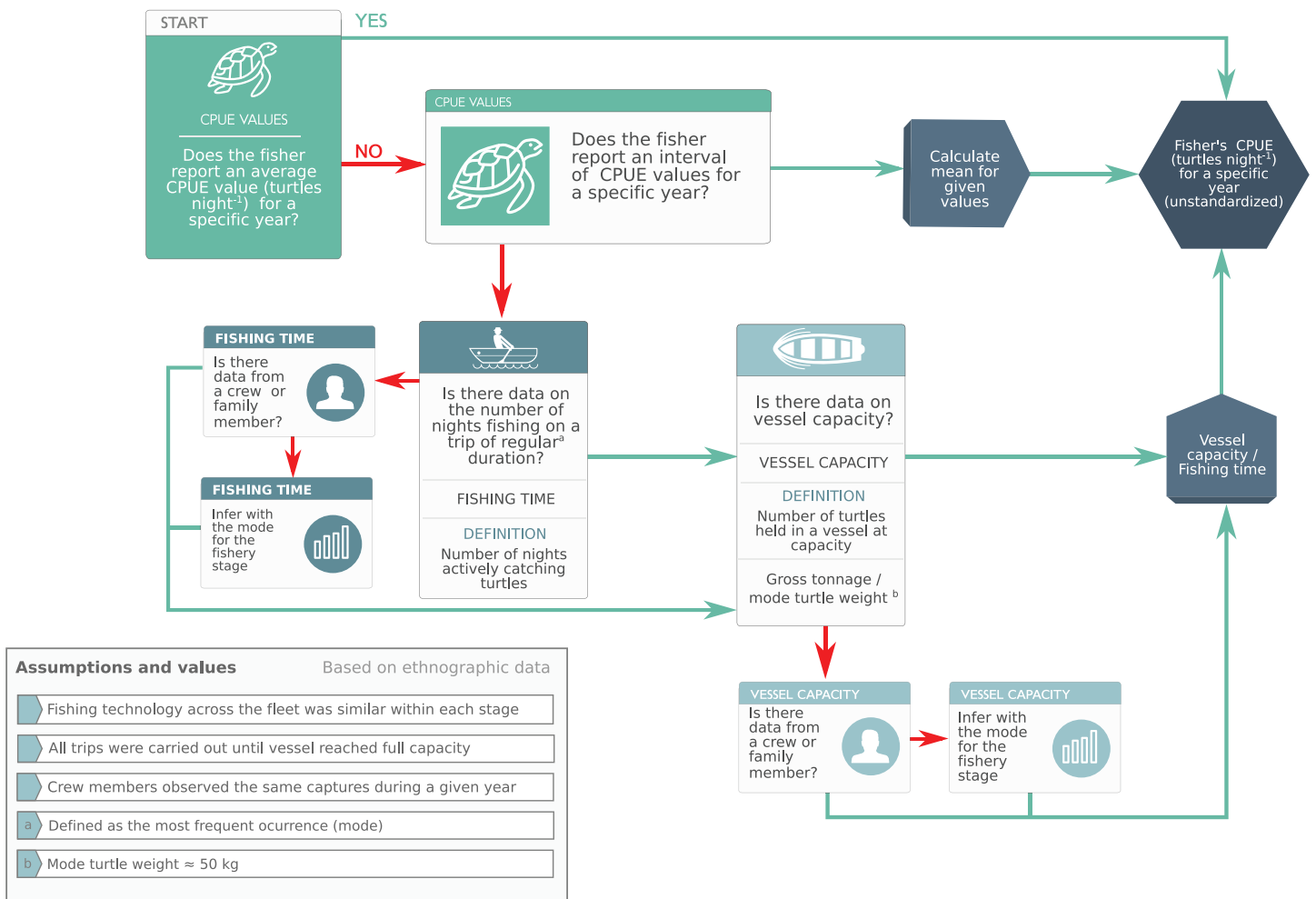
<sup>a</sup> Throughout the chronology, spatial distribution of fishing was highly variable due to the targeting of hot-spots and variations in the seasonal distribution of turtles.

<sup>b</sup> Size distribution was highly variable throughout the chronology.

Characteristics from qualitative LEK and *Early-Capistrán et al. (2018)*, *Márquez (1996)*, and *Seminoff et al. (2008)*.

young (10–13 years of age) when they captured turtles. During interviews, these fishers recognized that they had limited recollection of events and did not have the experience necessary to provide precise data. Statistical analysis confirmed that CPUE values provided by this group were outliers ( $\pm 2$  SD).

To evaluate CPUE trends, we converted values for the independent variable “year” to serial form in all analyses. We used *LABFit 7.2.49* to identify five preliminary models with best fit and their respective starting values. We then ran NLR (*nlstools*, *easynls*, *dplyr*, *car*, and *DescTools* packages; [Data and Code](#)) to choose the model that best described the data, and evaluated residuals ([Table 2](#)). We ran NLR at each round of the iterative process to (i) evaluate the general behavior and performance of the data, (ii) identify outlier effects in residual analysis, and (iii) evaluate if the process was robust to these effects



**Figure 3** Heuristic rules used to make systematic inferences based on expert knowledge to calculate raw catch-per-unit-effort values. We used heuristic rules to make systematic inferences based on the knowledge of expert turtle fishers (Phase 2, “CPUE Calculation and Preliminary Database Generation”). This framework allowed us to reduce individual cognitive bias by (i) complementing the knowledge of less experienced fishers with that of experts, and (ii) calculating a central tendency based on collectively-generated knowledge and biocultural consensus.

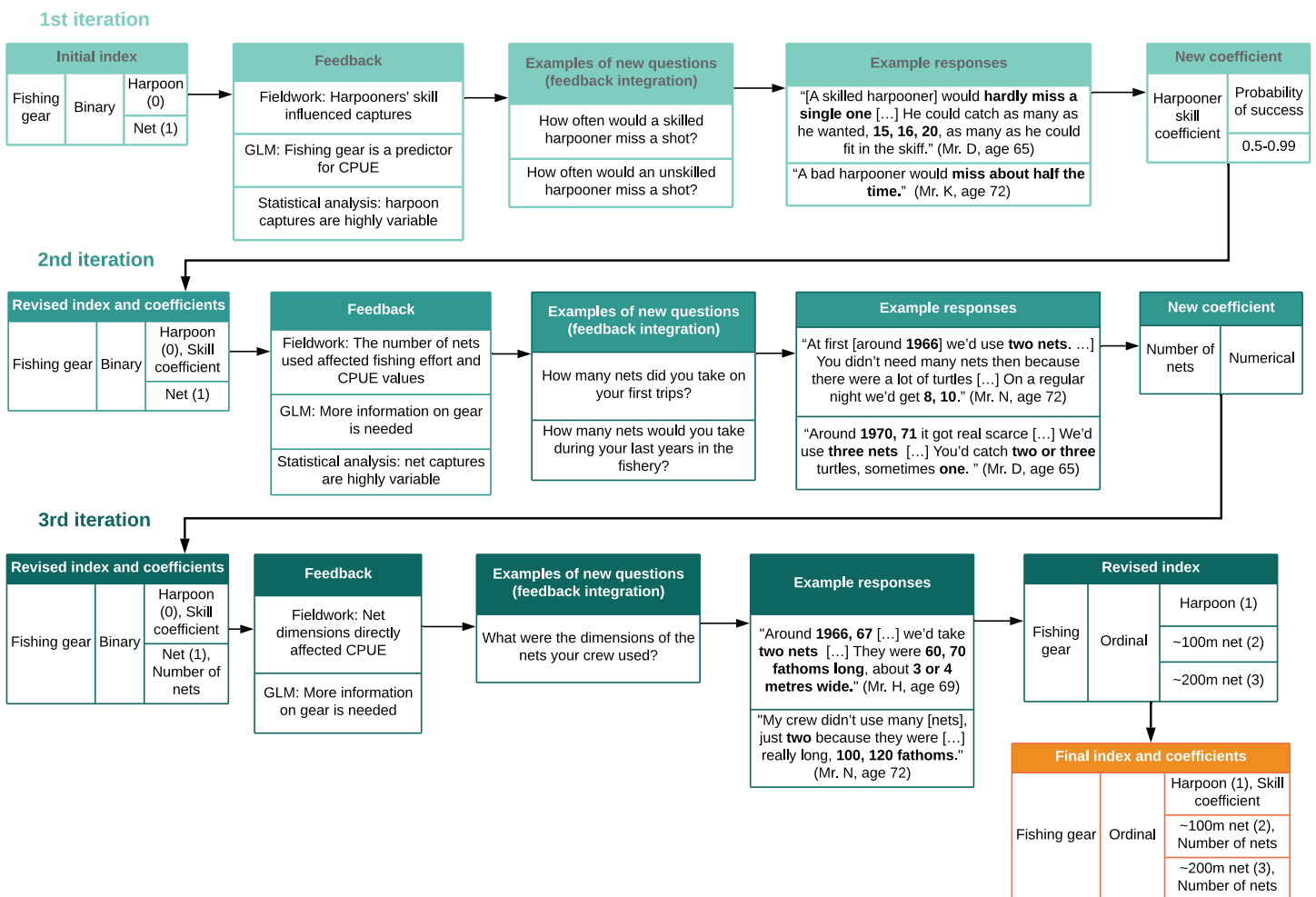
Full-size DOI: 10.7717/peerj.9494/fig-3

(Baty et al., 2015; Ritz & Streibig, 2008). An exponential decay model consistently showed the best fit, with the form

$$Y \sim \alpha \cdot e^{(\beta x)} \quad (2)$$

where  $Y$  is the response variable, CPUE;  $\alpha$  is a constant (intercept);  $\beta$  is an instantaneous rate of change in the response variable (slope); and  $x$  is the independent variable “year”.

We used GLMs with a link function for Gaussian distributions to identify significant predictor variables for CPUE (*nlme*, *dplyr*, *car* and *DescTools* packages), using log-transformed values if the CPUE distribution was non-normal (Zar, 2014). We ran backward-stepping models until we obtained a model with significant effects, a high percentage of explained deviance ( $D^2$ ), a relatively low Akaike Information Criterion (AIC), and robust residuals (Table 2) (cf. Maunder & Punt, 2004).



**Figure 4** Cyclical process of index design and feedback integration. We revised indices and coefficients based on a cyclical process which used feedback from interviews, statistical analysis, and generalized linear models (GLMs) to design new questions. This was repeated for each variable throughout Phase 2. Bold type shows numerical data from interviews. [Full-size !\[\]\(5f471a71b78d7676bc356df190b88ab4\_img.jpg\) DOI: 10.7717/peerj.9494/fig-4](https://doi.org/10.7717/peerj.9494/fig-4)

We ran a total of 36 NLR and 54 GLMs on five sequential working databases. The first three databases each corresponded to one round of the methodological cycle (Fig. 1). With each round, the working databases were updated and superseded as we incorporated new data, variables, indices, analyses, and data cleaning processes (Figs. 1, 3 and 4). The last two databases consisted of the final raw database—with LEK-derived CPUE values and heterogeneous variables for unit effort—and the standardized database with mean standardized CPUE values for each year (Data and Code). By integrating these analyses into the cyclical process, we are confident that we adequately identified confounding variables and sources of variation not attributable to changes in abundance (Hilborn & Walters, 1992).

### Feedback integration

We integrated model-fitting feedback by identifying which variables and indices required further information or could be improved (Fig. 4). We integrated feedback from

community members during subsequent visits to the field by sharing preliminary results and model outputs with them through narrative description, and asking for collaborators' perspectives on validity and consistency (Bélisle *et al.*, 2018). Local collaborators also identified gaps and provided further information (Huntington, 2000; Tengö *et al.*, 2014). We then designed new questions based on feedback and repeated these procedures with each variable (Figs. 1 and 4).

We repeated the cyclical process of data gathering, synthesis, and quantification until reaching topical saturation (similar instances were repeated and no additional data were found with which to develop new properties), thematic saturation (additional data did not produce new emerging themes), data saturation (new data repeated what was expressed in previous data) (Saunders *et al.*, 2018), and until model fitting did not provide significant new information.

Time frames required to reach saturation are extensive, in the order of months or years. Ethnographic fieldwork generally requires a year or more, given the extensive time required to establish rapport, obtain working knowledge and understanding of the cultural context, and to be able to ask good questions and obtain good answers (Bernard, 2011). The interview process to elicit the data presented in this article represented 57 working days over three field seasons (spring 2017, summer 2017, and spring 2018). While it may seem a rather short timeframe, it must be said that two of the authors, M.M.E.C. and G.G.M., have been conducting ethnographic work in the community since the summer of 2012, making seven trips to the region with a mean duration of 27 days, conducting a total of 378 interviews to date (Tables S2 and S3), and maintaining contact and communication with community members between field seasons. Long-term continuous interaction has allowed rapport for intelligible dialog among researchers and local community members in ways that enable elicitation of trustworthy data.

### **Phase 3: database standardization and validation**

#### ***Raw CPUE database analysis***

The result of the methodological cycle was a final, LEK-derived CPUE database with heterogeneous variables for unit effort (raw database) (Fig. 1). We carried out descriptive statistical analysis, NLR, and GLM analysis to evaluate the data and define standardization procedures as described in "Preliminary Data Evaluation".

#### ***CPUE database standardization***

We standardized CPUE to (i) remove most of the variation not attributable to changes in abundance by accounting for variables such as gears, fleet characteristics, fishers' experience, etc.; and (ii) generate CPUE values that could be compared over time (Hilborn & Walters, 1992; Maunder & Punt, 2004). To choose predictor variables for standardization, we ran GLMs (*nlme*, *car*, *dplyr*, and *DescTools* packages; Data and Code) with log-transformed CPUE values and a residual correlation structure based on an auto-regressive model of order 1 (AR-1) structured by the variable "year" (Zuur, 2009). We chose predictor variables for standardization using models with significant effects, high

percentage of explained deviance ( $D^2$ ), relatively low Akaike Information Criterion (AIC), and robust residuals (Table 2) (cf. [Maunder & Punt, 2004](#)).

We generated detailed definitions of unit effort based on these analyses, in order to obtain comparable values for turtles caught in one night. While fishers generally worked from dusk to dawn, fishing times on any given night with either gear type could be variable. For modeling purposes, we simplified values to 12 h blocks which reflect the vast majority of fishing effort (Article S1).

For set-nets, we standardized unit effort to approximate ecological monitoring data (100 m net soaking for 12 h) ([Koch, 2013](#); [Seminoff et al., 2003](#)):

$$C_{st} = (t \cdot R) / (n_r \cdot R \cdot 12 \text{ h}) \quad (3)$$

where  $C_{st}$  is a standardized, representative value of CPUE during a specific year (turtles  $12 \text{ h}^{-1}$ );  $t$  is the number of turtles caught (turtles); and  $n_r$  is the number of 100 m nets (no units).  $R$  is net length (in multiples of 100 m), simplified to short ( $\sim 100 \text{ m} = R$ ) or long ( $\sim 200 \text{ m} = 2R$ ) (Table 3). Soaking time is 12 h.

For harpoon captures, we assigned a skill coefficient ( $s$ , percentage of success) (Table 3) to each harpooner through social network analysis (Table 1), based on colleagues' assessment, such that:

$$C_{st} = t \cdot s^{-1} \cdot 12 \text{ h}^{-1} \quad (4)$$

The current ban on sea turtle fishing does not allow us to test for differences in susceptibility to fishing gears. Harpoons and nets were not used simultaneously by any given fisher, and both were used over a roughly equivalent number of hours per night. Thus, we considered these values to be adequately standardized given the nature of the data. For years with multiple CPUE values, we calculated the mean after standardization (Article S1; Figs. S4 and S5).

### **Evaluating statistical robustness**

We evaluated reliability through comparison with green turtle fishery statistics for BLA (annual landings in tons, 1962–1982) (Márquez cited in [Seminoff et al. \(2008\)](#)). CPUE and total landings are both crude indicators of abundance, and comparative analyses have been used to assess the accuracy of LEK-derived data ([De Damasio et al., 2015](#); [Sáenz-Arroyo & Revollo-Fernández, 2016](#)). We compared the catch reduction rate and fitted an exponential decay model (*QtiPlot 0.9.9.7*) as an experimental process to evaluate trends in LEK-derived CPUE and annual landings (Article S1). We then standardized both datasets to z-scores to avoid effects from differences in scales (Fig. S6) and used the Lin Concordance Correlation Coefficient (Lin CCC) to assess agreement between paired values (*DescTools* package; [Data and Code](#)) ([Lin, 1989](#); [Altman & Altman, 1999](#)) (Article S1; Fig. S6).

### **Phase 4: analysis of standardized CPUE data**

We performed descriptive statistical analysis and NLR on the standardized database, following the procedures described in “Preliminary Data Evaluation”, to understand



long-term abundance trends. We chose NLR for final analyses because (i) analyses conducted in Phases 2 and 3 consistently showed an exponential decay trend, and (ii) standardized CPUE data were not normally distributed (Shapiro–Wilk,  $p = 0.00334$ ), and NLR does not assume normally distributed data (Ritz & Streibig, 2008).

We ran a global sensitivity analysis using Markov-Chain Monte Carlo (MCMC) methods in R 3.4 (FME package; Data and Code) to derive the data-dependent probability distribution of the parameters. An MCMC samples from probability distributions to generate an ensemble of parameter values that represent the parameter distribution (Soetaert & Petzoldt, 2010). We ran an MCMC with initial values from the model with best fit ( $\alpha = 18.911$ ,  $\beta = -0.264$ ), with non-informative priors and 5000 iterations. We then estimated the effect of parameter uncertainty on the model output and generated a posterior predictive distribution of the model by taking a sample of the parameter probability function generated by the MCMC, running the model 100 times using a random draw of the parameters in the chain, and adding randomly distributed noise to estimate measurement error (Soetaert & Petzoldt, 2010).

## RESULTS

We generated a reliable, standardized green turtle fishery CPUE time-series from 1952 to 1982 by synthesizing and quantifying LEK. Three GLMs fit selection criteria to be used for database standardization, as described in “CPUE Database Standardization” (Table 2). These models showed that year, fishing gear type, vessel capacity, number of nets, net length, and fisher’s experience were significant predictor values for CPUE (Table 5). Given that each of these variables was incorporated into CPUE calculation and standardization, we are confident that both our estimates and standardization procedure were robust. Comparative analysis with fishery landing statistics confirmed accuracy: standardized CPUE and annual landings showed catch declines of 95% and 96%, respectively, and Lin CCC ( $\rho = 0.726$ ) showed strong agreement (Fig. 5).

All fishers consistently reported a declining trend during the fishery. This was consistent with our models. The NLR with best fit indicated that green turtle abundance declined exponentially through the four phases of the sea turtle fishery, likely driven by large-scale commercial exploitation with increased fishing effort and efficiency from 1960 to 1980 ( $R^2 = 0.798$ ) (Table 6; Fig. 6). Residual analysis suggested that the model was robust for the data (Table 2). MCMC sensitivity analysis suggested that our model is robust over parameter intervals for  $\alpha$  (15–35) and  $\beta$  (–0.12 to –0.06). Parameter values for the model with best fit ( $\alpha = 24.112$ ,  $\beta = -0.0929$ ) occurred within these intervals (Fig. S7). Furthermore, 94% of our data points occurred within the posterior predictive distribution, confirming that the model was a good fit for the data (Fig. S8).

## DISCUSSION

### Integrative methodological innovation

The importance of LEK data is increasingly recognized in conservation science (Lee et al., 2018; Mason et al., 2019). However, there has been reticence in the scientific community regarding the use of LEK due to concerns about accuracy, reliability, and potential biases

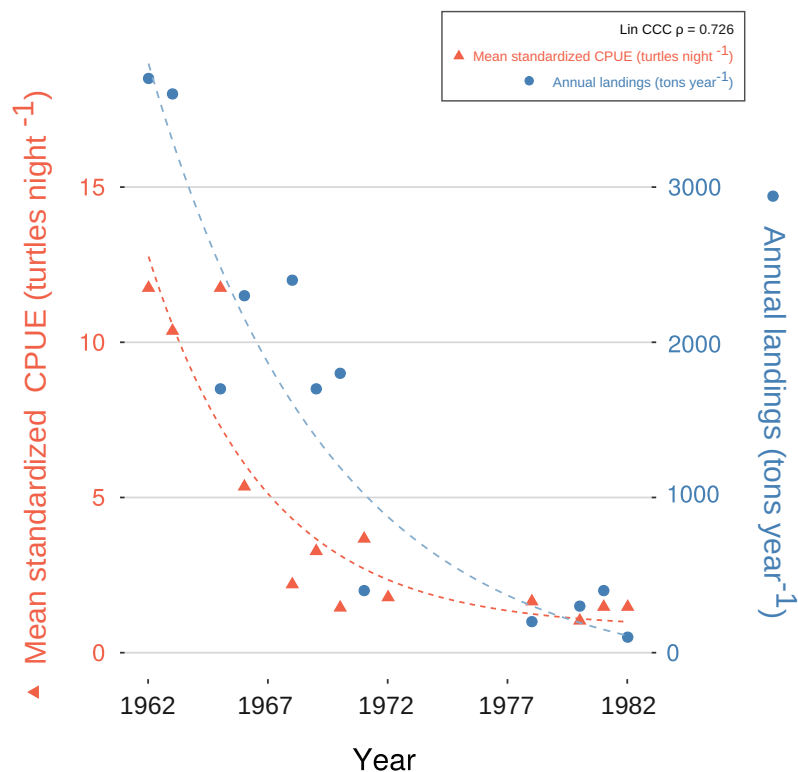
**Table 5** Generalized linear model (GLM) results for the raw catch-per-unit-effort (CPUE) database. The three most parsimonious GLMs for the raw database suggested that fishing gear type, vessel capacity, and number of nets were significant predictor variables for CPUE. Italics indicate significant results at  $\alpha = 0.95$ . Asterisks indicate significant result at  $\alpha = 0.90$ ; this variable was included to ensure robust residuals.

Predictors	Estimate	Std. error	P-value
Model 1: $\log(\text{cpue}) \sim \text{Year serial} + \text{Experience} + \text{Vessel Capacity} - 1$ ; AIC: 4.422; $D^2 = 0.775$ ; $df = 32$ ; $e_i \sim N(0, \sigma^2)$			
Correlation structure: Auto-regressive, Formula: $\sim \text{Year serial}$			
Year (serialised)	-0.278	0.00434	0.000
Experience	0.333	0.0328	0.000
Vessel Capacity	0.330	0.0692	0.000
Model 2: $\log(\text{cpue}) \sim \text{Year serial} + \text{Gear} + \text{Total Net Length} + \text{Number of Nets} + \text{Experience} - 1$ ; AIC: 10.215; $D^2 = 0.925$ ; $df = 20$ ; $e_i \sim N(0, \sigma^2)$			
Correlation structure: Auto-regressive, Formula: $\sim \text{Year serial}$			
Year (serialised)	-0.0239	0.0061	0.0014
Gear type	0.396	0.0980	0.0011
Total net length	-0.150	0.0429	0.0033
Number of nets	0.238	0.0750	0.0062
Experience	0.0969	0.0494	0.0689*
Model 3: $\log(\text{cpue}) \sim \text{Year serial} + \text{Gear} + \text{Net Length} - 1$ ; AIC: -11.710; $D^2 = 0.971$ ; $df = 32$ ; $e_i \sim N(0, \sigma^2)$			
Correlation Structure: Auto-regressive, Formula: $\sim \text{Year Serial}$			
Year (serialised)	-0.0284	0.00465	0.000
Gear type	1.324	0.0689	0.000
Net length	-1.321	0.0680	0.000

caused by differences in individual perception, memory, and recollection (O'Donnell, Pajaro & Vincent, 2010; Daw, Robinson & Graham, 2011). We are confident that such issues can be overcome through innovative, transdisciplinary methodologies that incorporate trusted methods from the social sciences and epistemological frameworks for incorporating multiple knowledge systems (Reed, 2008; Tengö et al., 2014; St. John et al., 2014). Our work contributes to overcoming key scientific challenges of using LEK as a consistent source of information by combining rigorous approaches to LEK documentation, synthesis, quantification, and statistical analysis.

We have approached the issues of accuracy, reliability, and recollection bias through several complementary processes. We used ethnography to document LEK, collecting verified, corroborated, and detailed qualitative and numerical data. Ethnographic data allowed for increased accuracy and reliability in comparison with data derived from structured questionnaire-based surveys or interviews alone. This particularly relevant in scenarios of high socio-environmental and biological complexity where multiple variables can affect or bias estimates of species abundance (Crandall et al., 2018; St. John et al., 2014). Ethnographic data also allowed us to understand the trajectory of human impacts on green turtle abundance in detail. This approach allowed us to describe, quantify, and integrate the social, economic, and technological processes that affected the green turtle





**Figure 5** Exponential decay model fitted to standardized catch-per-unit-effort (CPUE) values for *C. mydas* in Bahías de los Ángeles, derived from local ecological knowledge (LEK). Data points are mean, standardized LEK-derived CPUE values for a specific year (red triangles and dotted line; left Y-axis) and total annual landings from available fisheries statistics for Bahía de los Ángeles (blue circles and dotted line; right Y-axis) (Márquez in *Seminoff et al. (2008)*). Curves represent suggested trends based on an exponential decay model (details in *Article S1*). Lin Concordance Correlation Coefficient of paired z-scores suggests strong agreement between datasets (see also *Fig. S6*).

Full-size DOI: 10.7717/peerj.9494/fig-5

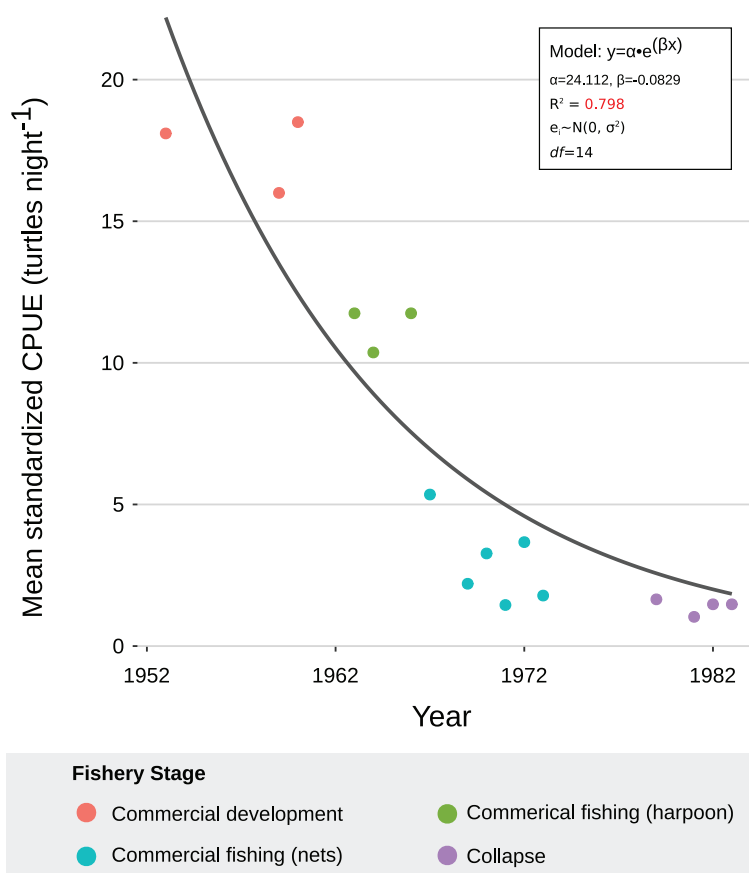
**Table 6** Results of nonlinear regression with best fit for catch-per-unit-effort estimates derived from local ecological knowledge. Italics indicate significant results at  $\alpha = 0.95$ . See also *Fig. 6*.

Parameter	Estimate	Std. error	95% CI	t-value	P-value
$\alpha$	24.112	3.124	[17.413–30.812]	7.719	<i>2.07e-06</i>
$\beta$	-0.0829	0.0130	[-0.111 to -0.0551]	-6.382	<i>1.71e-05</i>

Note:

Model:  $Y \sim \alpha \cdot e^{(\beta x)}$ ;  $df = 14$ ;  $e_i \sim N(0, \sigma^2)$ .

fishery into our estimates and indices (e.g., changes in fishing gear and displacement capacities, commercial demand, spatial dynamics, etc.). Thus, we incorporated detailed knowledge of the nature and evolution of the green turtle fishery into our models. We reduced cognitive bias and recollection bias by estimating CPUE on the basis of biocultural consensus from multiple, nested knowledge systems rather than direct individual recollection. Finally, we integrated statistical analysis and feedback throughout all phases of our methodology to assure statistical robustness.



**Figure 6** Exponential decay model fitted to mean, standardized catch-per-unit-effort (CPUE) values for *C. mydas* in Bahía de los Ángeles, derived from local ecological knowledge (LEK). Curve represents the nonlinear regression with best fit and robust residuals, based on an exponential decay model. Each data point is a representative, mean, standardized CPUE value for a specific year derived from LEK data. Colors represent fishery stages (see Table 4). Parameter values, standard error, confidence intervals, *t*-values, and *P*-values can be found in Table 6. [Full-size !\[\]\(1663bb69f307a960345edb0e712f8c02\_img.jpg\) DOI: 10.7717/peerj.9494/fig-6](https://doi.org/10.7717/peerj.9494/fig-6)

The strong concurrence of our LEK-derived CPUE estimates with fishery landing data for the historical fishery years (1962–1982) helps confirm the accuracy of LEK as a source of information for understanding population trends in the recent past (De Damasio *et al.*, 2015; Sáenz-Arroyo & Revollo-Fernández, 2016). Robust model-fitting and sensitivity analyses confirmed statistical reliability. Thus, we are confident that our methods provide practical approaches to the scientific challenges of using LEK in conjunction with ecological modeling through detailed LEK documentation, biocultural consensus, and continuous statistical analysis and feedback.

Spatial dynamics present an interesting area of opportunity for future research building upon our methods. The development of sound approaches to management and conservation requires understanding fishing effort over both space and time (Anticamara *et al.*, 2011; Selgrath *et al.*, 2018). We approached spatial variability through proxies (e.g., propulsion and trip duration, Tables 3 and 4) due to the very high variability and complexity of spatial dynamics over time. Given that our primary focus was on temporal

trends in CPUE, these proxies provided a simple index to account for spatial effort (Hilborn & Walters, 1992; Walters, 2003). LEK-based studies of spatial dynamics have primarily been conducted with present-day, data-poor fisheries (cf. Moore et al., 2010; Moreno-Báez et al., 2010; Selgrath et al., 2018). In future, the application of our methods for detailed reconstruction of the spatial dynamics of the green turtle fishery through LEK could have important potential for uncovering historical green turtle spatial distribution and habitat use, and understanding changes in spatial dynamics over broad temporal scales.

We recognize that LEK data is epistemologically distinct from technical data, and have aimed to bridge epistemological gaps and produce a synergistic integration of LEK and scientific methods (cf. Brook & McLachlan, 2005; Tengö et al., 2014). As scientists, we recognize that our research is value-laden and that the inevitable differences between LEK and technical data are more often reflections of epistemological differences or methods of collection than inherent unreliability (Brook & McLachlan, 2005). LEK research requires trust-based collaboration between researchers and communities, a process that can necessitate years of commitment (Brook & McLachlan, 2005). In such contexts, when researchers can elicit and corroborate qualitative data derived from empirically-lived situations (Palmer & Wadley, 2007), synthesize and quantify this data, and submit quantified data to rigorous mathematical analysis, they can assure that LEK-derived estimates are accurate and statistically reliable. Such information is of crucial importance for conservation and management, particularly in scenarios where there is a need for understanding long-term trends; where technical data are scarce or unavailable; or where species are impacted by illegal, unregulated or undocumented exploitation (Duffy et al., 2016; Pauly, 1995; Sáenz-Arroyo & Revollo-Fernández, 2016). Concomitantly, the integration of LEK and scientific knowledge offers the possibilities of incorporating and empowering local conservation processes with peoples previously seen as deleterious agents for those same environments and species of which they hold a vast amount of LEK (cf. Berkes et al., 2005). Lastly, the use of LEK provides comprehensive understanding of complex and dynamic socio-ecological processes while facilitating the creation and implementation of culturally appropriate local solutions to environmental problems (cf. Reed, 2008; Brown, 2010).

### Understanding east pacific green turtle population trends

Our LEK-derived CPUE data provide a baseline abundance of green turtles before large-scale commercial exploitation at a key feeding area in the Gulf of California, and describe population trends prior to ecological monitoring which are essential for establishing conservation and management goals (McClenachan et al., 2016; Seminoff et al., 2003). Our approach provides a historical reference point for the Bahía de los Ángeles foraging population and enables us to better understand contemporary datasets and current population status in the area (Seminoff et al., 2008). Our results suggest that fishery-derived mortality exceeded replacement via reproduction or immigration rates into the feeding areas (Chaloupka & Musick, 1996). Furthermore, although fishing effort and efficiency increased over time, previous CPUE could not be maintained due to the

overall decline in green turtle abundance (*Hilborn & Walters, 1992*). We are confident that CPUE values in the 1950s can be considered an adequate historical baseline abundance level, based on previous research which identified the early 1960s as a period when human impacts precipitated a major decline in green turtle abundance in the Gulf of California (*Early-Capistrán et al., 2018*).

Future research that pairs LEK-derived estimates with contemporary in-water monitoring and nesting data can provide fundamental insights for conservation status evaluations such as those conducted under the auspices of the IUCN Red List (*Mazaris et al., 2017; Seminoff & Shanker, 2008*). Such long-term perspectives are generally not attainable via scientific monitoring efforts alone, especially considering that although sea turtles have been exploited worldwide for centuries or millennia, even the longest tenured sea turtle monitoring programs only started in the 1960s (*Balazs & Chaloupka, 2004; Bjørndal, Bolten & Chaloupka, 2005; Chaloupka & Limpus, 2001; Márquez, 1996*).

In the case of BLA, existing baseline data from 1995 correspond to a decimated population, and would thus be prone to over-estimating the degree of initial recovery observed from the early 2000s onward (*Delgado-Trejo, 2016; Pauly, 1995; Seminoff et al., 2015*). Currently, scientific surveys are conducted monthly using CPUE as an index. Catch effort is variable within specific parameters, using 100–200 m set-nets and 12–24 h soak times (*Koch, 2013; Seminoff et al., 2008*). In future, our standardized LEK-derived CPUE estimates can be integrated with standardized monitoring data to provide a long-term view of green turtle abundance at this index feeding area. Integration of past trends with modern-day survey data is crucial for evaluating the overall conservation status of the East Pacific green turtle with references to baseline abundance levels prior to large-scale commercial exploitation (*Broderick et al., 2006; Seminoff & Shanker, 2008; Wildermann et al., 2018*).

## CONCLUSIONS

Our reconstruction of baseline conditions revealed an exponential decline in green turtle abundance between 1960 and 1980 at Bahía de los Ángeles, one of the most important and productive green turtle commercial fishing areas in the eastern Pacific Ocean (*Caldwell, 1963; Early-Capistrán et al., 2018*). As scientific monitoring began only in 1995 after population collapse, no pre-exploitation baseline data were available to evaluate current abundance and conservation status (*Seminoff et al., 2008*). Our LEK-derived data can now provide historical context and a reliable baseline abundance estimate for this green turtle population. We are confident that future studies integrating our LEK-derived estimates with current scientific monitoring data from both foraging habitats and nesting beaches will yield a holistic, long-term perspective of green turtle abundance, conservation, and population dynamics in the eastern Pacific.

Beyond reconstructing green turtle abundance, our methodology may be exported to parallel cases dealing with the conservation and monitoring of other long-lived species that are fished as it can unravel complex phenomena by combining LEK and ecological modeling. We provide a framework to overcome the challenges of documenting and quantifying LEK, and bridge practical and epistemological gaps (*Mistry & Berardi, 2016*;

*Mukherjee et al., 2018*). This approach provides a way to deal with variation in individual memory using corroborated data and collectively produced knowledge, to simplify and manage large volumes of qualitative information, and to translate qualitative data into a format compatible with ecological modeling (*Bélisle et al., 2018*).

We recognize that of LEK and derived population abundance estimates are technically and epistemologically distinct from data obtained under experimental conditions. Nevertheless, they can provide a robust description of significant inflection points in abundance trends that would be less-resolved if analyses were limited to scantily-available technical data (*Pauly, 1995; Sáenz-Arroyo & Revollo-Fernández, 2016*). LEK-based and integrative approaches can provide long-term information where scientific monitoring data are scarce or unavailable, and contribute to collaborative knowledge production (*Barrios-Garrido et al., 2018; Lee et al., 2018; Mistry & Berardi, 2016*). While our methods are most readily adapted to marine fauna such as marine mammals, reptiles, teleost fish, and long-lived invertebrates, this approach can also be modified and applied to terrestrial and freshwater biota. We trust that future research that rigorously integrates social and ecological science can help address challenges for conservation and management in the context of global change and biodiversity loss (*Mukherjee et al., 2018; Sutherland et al., 2018*).

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Volker Koch is working as a policy advisor at Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) supporting the Federal Ministry for Economic Cooperation and Development on topics relating to sustainable chemicals management and Ecological Fiscal Reform.

### Author Contributions

- Michelle-María Early-Capistrán conceived and designed the experiments, performed the experiments, analyzed the data, prepared figures and/or tables, authored or reviewed drafts of the paper, conducted ethnographic fieldwork and interviews, and approved the final draft.
- Elena Solana-Arellano conceived and designed the experiments, analyzed the data, authored or reviewed drafts of the paper, supervised research, and approved the final draft.
- F. Alberto Abreu-Grobois conceived and designed the experiments, authored or reviewed drafts of the paper, supervised research, and approved the final draft.
- Nemer E. Narchi conceived and designed the experiments, prepared figures and/or tables, authored or reviewed drafts of the paper, supervised research, and approved the final draft.
- Gerardo Garibay-Melo performed the experiments, authored or reviewed drafts of the paper, conducted ethnographic fieldwork and interviews, and approved the final draft.
- Jeffrey A. Seminoff conceived and designed the experiments, authored or reviewed drafts of the paper, and approved the final draft.
- Volker Koch conceived and designed the experiments, authored or reviewed drafts of the paper, and approved the final draft.
- Andrea Saenz-Arroyo conceived and designed the experiments, authored or reviewed drafts of the paper, and approved the final draft.

### Human Ethics

The following information was supplied relating to ethical approvals (i.e., approving body and any reference numbers):

Research was approved by the Bioethics Committee of the Centro de Investigación Científica y de Educación Superior de Ensenada (Approval Number 2S.3.1).

### Data Availability

The following information was supplied regarding data availability:

Quantitative datasets without personally identifiable information and R code are available as a [Supplemental File](#) and and GitHub (<https://github.com/earlycapistran/PeerJ>).

Due to the sensitive and confidential nature of ethnographic data, all primary ethnographic data (including field notes, field journals, photographs, archival materials, audio and video recordings, and transcriptions) are stored in secure archives and may only be accessed by the core research team (Michelle-María Early-Capistrán, Elena Solana-Arellano, and F. Alberto Abreu-Grobois), in compliance with the ethical guidelines of the International Society of Ethnobiology.

## Supplemental Information

Supplemental information for this article can be found online at <http://dx.doi.org/10.7717/peerj.9494#supplemental-information>.

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## **ARTÍCULO 2: INTEGRACIÓN DEL MONITOREO ECOLÓGICO Y EL CONOCIMIENTO ECOLÓGICO LOCAL PARA EVALUAR LOS RESULTADOS DE LA CONSERVACIÓN**

### **Resumen**

La conservación exitosa de especies longevas requiere el entendimiento de las tendencias a largo plazo y las líneas de base históricas. A través de un estudio de caso una agregación de alimentación de tortuga prieta (*Chelonia mydas*) en el norte del Golfo de California, México, integramos datos de monitoreo científico con reconstrucciones históricas de tasas de captura derivadas del Conocimiento Ecológico Local (LEK, por sus siglas en inglés). Los modelos ajustados a datos de LEK y monitoreo indican que la abundancia de tortugas está aumentando, pero solo después de ~40 años de salvaguardar los hábitats de anidación y alimentación de esta especie en México. Sin embargo, la población disminuyó un 75% más rápido de lo que ha aumentado, y la abundancia actual se encuentra en ~60% de los niveles de referencia históricos, lo cual indica que es necesario continuar con acciones de conservación sostenidas a largo plazo. Este estudio demuestra el potencial de la vinculación LEK y la ciencia ecológica para proporcionar información crítica para la conservación mediante el establecimiento de líneas de base y la evaluación del estado poblacional, a la par con el fomento de futuros equitativos y sostenibles para las comunidades locales.

## Sobretiro del artículo 2

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1 **Integrating ecological monitoring and local ecological knowledge to**  
2 **evaluate conservation outcomes**

3

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22

23

24 **Keywords (10 max.):** Conservation, local ecological knowledge, long-term data, marine

25 historical ecology, non-linear modeling, sea turtles, transdisciplinary research

26

## 27 **Abstract**

28 Successful conservation of long-lived species requires reliable understanding of long-term  
29 trends and historical baselines. Using a green turtle (*Chelonia mydas*) foraging aggregation  
30 in the northern Gulf of California, Mexico as case study, we integrated scientific  
31 monitoring data with historic catch rate reconstructions derived from Local Ecological  
32 Knowledge (LEK). Models fit to LEK and monitoring data indicate that turtle abundance is  
33 increasing, but only after ~40 years of safeguarding the species' nesting and foraging  
34 habitats in Mexico. However, as population declines occurred 75% faster than increases,  
35 and current abundance is at ~60% of historical baseline levels, indicating the need for  
36 sustained, long-term conservation actions. This study demonstrates the potential of linking  
37 LEK and ecological science to provide critical information for conservation, by establishing  
38 reference baselines and gauging population status, while promoting equitable and  
39 sustainable futures for local communities.

40



## 41 **Introduction**

42 Successful conservation of highly migratory, long-lived species such as sea turtles requires  
43 efforts and policies implemented across large spatio-temporal scales, along with locally-  
44 grounded data and practices (Mazaris et al. 2017; Vierros et al. 2020). Local Ecological  
45 Knowledge (LEK) can contribute to sound conservation and management practices by  
46 establishing local baselines and recovery targets; evaluating population status; integrating  
47 the cultural dimensions of human-environment interactions; and supporting equitable and  
48 inclusive practices (Early-Capistrán et al. 2020a; Lee et al. 2018; Poe et al. 2014). LEK-  
49 based approaches are particularly important for taxa impacted by small-scale fisheries or  
50 subsistence hunting, for which technical or baseline abundance data may be scarce or un-  
51 available (cf. Sáenz-Arroyo & Revollo-Fernández 2016; Selgrath et al. 2018).

52 East Pacific green turtles (henceforth, green turtles) are a regionally-distinct  
53 population segment of the cosmopolitan green turtle (Seminoff et al. 2015). These marine  
54 megaherbivores are ecosystem modifiers, with vital roles in marine food-web and  
55 ecosystem dynamics (cf. Christianen et al. 2021). They are highly migratory and occupy a  
56 broad range of habitats across different life stages, primarily tropical reproductive areas and  
57 tropical or warm-temperate foraging areas separated by hundreds or thousands of  
58 kilometers (cf. Seminoff et al. 2015). Globally, monitoring is heavily skewed towards  
59 nesting habitats where only breeding females and egg output are quantified (cf. Seminoff &  
60 Shanker 2008). However, coastal and neritic foraging habitats incorporate a broad range of  
61 turtle age classes and both sexes, providing insights into population dynamics and trends  
62 crucial for informing conservation policy (Bjorndal et al. 2005; Seminoff & Shanker 2008;  
63 Seminoff et al. 2003).

64 Like other sea turtle species, green turtles are highly susceptible to over-exploitation  
65 due to their complex life-history and prolonged maturation time. Worldwide, the long  
66 history of commercial exploitation of meat and eggs has depleted or extirpated several  
67 green turtle stocks (cf. Chaloupka et al. 2008). The Baja California peninsula provides a  
68 unique case study with a robust historical baseline within living memory (the 1950s)  
69 (Early-Capistrán et al. 2018). Green turtles are a cultural keystone species (Garibaldi &  
70 Turner 2004) used over millennia by the region's inhabitants, with fundamental roles as  
71 food and medicine. Small human populations and geographic isolation kept captures  
72 sustainable until the mid-twentieth century. From 1960-1980, intensive commercial fishing  
73 supplied green turtle meat to fast-growing cities along the U.S.-Mexico border, driving  
74 foraging populations to near extinction (Early-Capistrán et al. 2018). This process was  
75 coupled with intense egg collection in the populations' tropical reproductive habitats  
76 (Delgado-Trejo & Alvarado Díaz 2012; Seminoff et al. 2015) (Figure 1). Long-term  
77 conservation and research began in the late 1970s as green turtles diminished, and in the  
78 early 1980s the fishery collapsed (Early-Capistrán et al. 2020a; Seminoff et al. 2008)  
79 (Figure 2).

80 Local, regional, and international conservation efforts, including nesting beach and  
81 habitat protection, bans or restrictions on direct use, and by-catch regulations, have  
82 successfully reversed declines in some populations, including green turtles in the east  
83 Pacific (Broderick et al. 2006; Mazaris et al. 2017). Green turtles are classified as  
84 endangered under Mexican law and by the IUCN, and all sea turtle exploitation in Mexico  
85 has been banned since 1990 (Diario Oficial de la Federación 1990; IUCN 2019). Decades

86 of nesting beach and habitat protection across the species' range in Mexico have led to  
87 population increases since the early 2000s (cf. Seminoff et al. 2015).

88 We integrated scientific monitoring data with historical catch rate reconstructions  
89 generated collaboratively with local fishers by documenting and quantifying LEK, resulting  
90 in (to our knowledge) the longest available standardized time-series (1952-2018)  
91 worldwide for a sea turtle foraging habitat. We aim to understand long-term change,  
92 contextualize current local abundance compared to historical baselines, and help establish  
93 future directions for conservation and management policies. Our innovative, long-term  
94 approach can help link ecological science and LEK, promoting conservation processes that  
95 harness the collective capacity of local and scientific communities (Game et al. 2015).

96

## 97 **Methods**

### 98 *2.1 Study site*

99 Bahía de los Ángeles (BLA), Baja California, Mexico (28.951917°, -113.562433°) is a  
100 warm-temperate green turtle foraging area in the Gulf of California (Figure 1). This index  
101 site hosts significant in-water aggregations of turtles and systematic long-term scientific  
102 monitoring (Early-Capistrán et al. 2020a), initiated in 1979 through government-sponsored  
103 efforts (Seminoff et al. 2008). In-water monitoring began in 1995 and has continued as a  
104 joint effort between government, academic, and non-governmental institutions (Figure 2).  
105 Green turtles do not nest in BLA. The index nesting beach for this population is in Colola,  
106 Michoacán, Mexico, ~1500 km southeast of BLA (18.297392°, -103.410956°) (Seminoff et  
107 al., 2003). Establishing direct connectivity with stock-specific nesting rookeries is

108 challenging, as green turtle foraging areas incorporate individuals from multiple genetic  
109 stocks (Dutton et al. 2019; Seminoff et al. 2015). However, Colola is the only green turtle  
110 nesting site in the Northeast Pacific with long-term data (>30 years), and accounts for 56-  
111 71% of green turtles in the Gulf of California's foraging areas (Delgado-Trejo 2016; Koch  
112 2013).

113

## 114 ***2.2 LEK-derived data (1952–1982): commercial fishing***

115 We used an existing dataset of LEK-derived catch-per-unit-effort (CPUE) estimates to  
116 establish baseline abundance and analyze long-term change before scientific monitoring  
117 (Early-Capistrán et al. 2020b) (Figure 2). We compiled data in an on-going  
118 transdisciplinary research process with the BLA community initiated in 2012. We  
119 documented, corroborated, and classified LEK through ethnography; synthesized through  
120 coding, indexing, and heuristics; and integrated feedback processes with statistical  
121 analyses. We standardized all CPUE estimates to one 12-h in-water set for a single 100m  
122 net. Standardization (i) removed the majority of variation not attributable to abundance  
123 changes (e.g., fishing gear types, fleet characteristics, spatial dynamics); and (ii) generated  
124 estimates compatible with monitoring data. To evaluate central tendencies, we used annual  
125 mean CPUE for all analyses (Supporting Information: 1, Figure S1, Table S1). LEK data  
126 were shown to be statistically reliable and generated robust models of population change  
127 (Early-Capistrán et al. 2020a). See Early-Capistrán and collaborators (2020a) for detailed  
128 methodological processes.

129

### 130 **2.3 Scientific monitoring (1995-2018): conservation**

131 In-water monitoring began in 1995 after population collapse. Using CPUE as an index,  
132 turtles were captured with set-nets of the same design as those used in commercial green  
133 turtle fishing (Seminoff et al. 2003) (Supporting Information: 2). Monitoring data were  
134 provided by author J.A. Seminoff (2003; NOAA, Unpublished raw data) for 1995–2005  
135 and Comisión Nacional de Áreas Naturales Protegidas (CONANP) & Grupo Tortuguero de  
136 Bahía de los Ángeles (Unpublished raw data) for 2005–2018. We ensured direct  
137 compatibility with LEK data by standardizing to one 12-h in-water set for a single 100m  
138 net (Seminoff et al. 2003) and analyzing mean annual CPUE values (Supporting  
139 Information: 1; Figure S1; Tables S1, S2).

140

### 141 **2.4 Analyses**

#### 142 *Multiple Imputation by Chained Equations (MICE)*

143 We evaluated CPUE trends for two distinct periods: Commercial Fishing (1952–1983) and  
144 Conservation (1978–2018) (Figure 2). For the Conservation period, we appended LEK  
145 values from 1978–1982 to the monitoring dataset and interpolated values in the temporal  
146 gap between the end of commercial fishing (1983) and the start of in-water monitoring  
147 (1995). This is justified, as conservation efforts began in the late 1970s and increased over  
148 time (Figure 2) (Márquez 1996; Seminoff et al. 2008) (Supporting Information: 3).

149 We ran multiple imputation by chained equations (MICE) with the *mice* package *in*  
150 *R 4.0.4* (van Buuren & Groothuis-Oudshoorn 2011) to handle the high percentage of  
151 missing values (Commercial Fishing = 48%, Conservation = 42%). This systematic missing

152 data method is robust for scenarios with up to 75% missingness (Takahashi 2017) (Figure  
153 3). We assumed that values were missing at random (i.e., missingness was not  
154 systematically related to catch rates) (Thurstan et al. 2014).

155 Each missing annual value was replaced with a plausible value generated using a  
156 Markov Chain Monte-Carlo algorithm to sample subsets of observed values (van Buuren &  
157 Groothuis-Oudshoorn 2011) (Supporting Information: 4.1). We generated  $m$  complete  
158 datasets, equivalent to the percentage of missing values for each phase, by running 1500  
159 iterations (Bodner 2008) (Figures S2, S3). Observed values were retained.

160 We fitted each of the  $m$  complete datasets separately to the model:

$$161 \quad Y \sim \alpha \cdot e^{(\beta x)} \quad (\text{eqn. 1})$$

162 Where  $Y$  is the response variable (CPUE),  $x$  is the independent variable (year), and  $\alpha$  and  $\beta$   
163 are fitted constants. MICE does not generate a singular regression. Instead, the results of  
164 the  $m$  fitted models are pooled using Rubin's Rules (Dong & Peng 2013) to give parameter  
165 estimates with standard errors that (i) describe the uncertainty of imputed missing data; (ii)  
166 account for variance within and between imputed models; and (iii) are unbiased and have  
167 valid statistical properties (van Buuren & Groothuis-Oudshoorn 2011) (Figure 3)  
168 (Supporting Information: 4.2, 4.3; Table S3).

169 We ratified validity through residual analysis ( $e_i \sim N(0, \sigma^2)$ ) (Nguyen et al. 2017)  
170 (Supporting Information: 4.4). We used an *ad hoc* method to (i) visualize a pooled trend  
171 line (broadly equivalent to regression line), showing pooled predicted values across all  $m$   
172 multiply imputed models, and (ii) draw 95% Confidence Intervals for the pooled trend line

173 using Rubin's Rules (Dong & Peng 2013; Nguyen et al. 2017) (Supporting Information:  
174 4.5).

175

### 176 *Population growth and size distribution*

177 We calculated population growth rates during Commercial Fishing (1953–1982) and  
178 Conservation (1978–2018) (Supporting Information: 5). Curved carapace length (CCL) size  
179 distributions from monitoring data were converted to life stages based on mean nester size  
180 at Colola (82.0 cm CCL) (Seminoff et al. 2015); i.e. adults > 82.0. We used Mann-Whitney  
181 U tests ( $\alpha = 0.05$ ) to compare mean annual catch rates in the LEK and monitoring datasets  
182 (Supporting Information: 6, 7), and to compare size and life stage composition over time  
183 (Period 1: 1995–2005; Period 2: 2009–2018).

184

## 185 **Results**

186 Our results strongly suggest increasing green turtle abundance after ~40 years of  
187 conservation measures and ~30 years of full legal protection (Figure 4a). Both processes,  
188 Commercial Fishing and Conservation, are described by a nonlinear model (eqn. 1). The  
189 Commercial Fishing period has a high  $\alpha$  and negative  $\beta$  ( $\alpha = 24.271$ ,  $\beta = -0.820$ ;  $R^2 =$   
190  $0.845$ ) while during the Conservation period a low  $\alpha$  and positive  $\beta$  ( $\alpha = 0.002$ ,  $\beta = 0.136$ ;  
191  $R^2 = 0.711$ ), both with robust residuals and good fits (Figure 5; Table 1).

192 The BLA population declined at a rate of 8.4% annually during commercial fishing,  
193 which contrasts with the 4.8% annual increase during the conservation phase (Table S4),

194 emphasizing that declines occurred 75% faster than increases. Notably, there is a prolonged  
195 latency period between the initial implementation of conservation measures (1979), initial  
196 signs of increase (~2000), and the significant recovery phase after 2011. Despite the clear  
197 upward trend, maximum CPUE during scientific endeavor (11.2 turtles/12hr; 2018)  
198 represents ~60% of the maximum CPUE in the commercial fishery (18.5 turtles/12hr;  
199 1959) (Figures 4a, 5). Furthermore, median CPUE is significantly lower in scientific  
200 monitoring (Median = 0.66) than commercial fishing (Median = 3.47) (Mann-Whitney U =  
201 232,  $p < 0.05$ , 95% C.I. [0.81–7.75]) (Tables S2, S5). Thus, although abundance levels in  
202 BLA are near those of the mid-1960s, they still remain below the historical baseline.

203

#### 204 *Life stage and size distribution (1995–2018)*

205 Abundance increases after 2011 coincide with shifts toward a higher proportion of juvenile  
206 turtles (Figure 4b). Period 2 (2009–2018; 72.22% juveniles) shows a strong juvenile bias  
207 compared with Period 1 (1995–2005) (55.71% juveniles). Furthermore, median CCL was  
208 significantly smaller in Period 2 (Median = 75.5) than Period 1 (Median = 80.8) (Mann-  
209 Whitney U = 71406,  $p < 0.001$ , 95% C.I. [1.99–5.20]) (Tables S6, S7). These patterns  
210 suggest juvenile recruitment drives population growth and are consistent with sea turtles'  
211 prolonged somatic growth rates, which require decades of hatchling production before  
212 juvenile recruitment into foraging habitats is observed (Seminoff & Shanker 2008).

213

## 214 **Discussion**



215           The robust positive trend for foraging populations at BLA shows that conservation  
216 measures work, but require prolonged time frames. While intensive, technologically  
217 efficient commercial fishing—even by a small artisanal fleet— can have severe impacts in  
218 a relatively short time (Peckham et al. 2007), population recovery requires decades of  
219 sustained protection across habitats. In our case, whereas collapse occurred in ~20 years,  
220 significant population growth was only possible after ~40 years of conservation. This  
221 pattern of fast decline and slow population growth is consistent with other long-lived  
222 marine taxa, including large sharks, sirenids, and cetaceans (cf. Chaloupka et al. 2008;  
223 McClenachan et al. 2012).

224           Our results further demonstrate that for depleted long-lived, highly migratory  
225 species like sea turtles, spatially and temporally widespread conservation measures—  
226 nesting beach and habitat protection, by-catch regulation, and protection from unsustainable  
227 use of meat or eggs—are pre-requisites for positive results (Chaloupka et al. 2008). The  
228 BLA foraging population shows encouraging trends. Nonetheless, current local abundance  
229 remains significantly below baseline levels. This result is consistent with the consensus  
230 among senior fishers that green turtles are abundant but below numbers observed when  
231 they were young harpooners (Early-Capistrán et al. 2020a). Thus, continued conservation  
232 measures are required for the population to return to historical levels.

233

### 234 *The need for integrative analysis*

235           The evaluation of green turtle conservation status is complicated by the lack of  
236 knowledge about fundamental parameters, including maturation age, life-stage duration,

237 and migratory and demographic connectivity (Casale & Heppell 2016; Seminoff & Shanker  
238 2008). Holistic assessment is beyond the scope of this paper. However, population  
239 trajectories at BLA largely coincide with trends at the Colola index nesting site. The Colola  
240 coastline was largely uninhabited until the 1950s. Settlements grew as market demand for  
241 sea turtle products surged in the early 1970s, and ~70,000 eggs/night were harvested  
242 commercially until nesting beach protection began in 1979 (Delgado-Trejo & Alvarado  
243 Díaz 2012). Like BLA's foraging population, the rookery at Colola has grown substantially  
244 since ~2010: nesting rates increased 508% from 1982 (3,383 nests/year) to 2015 (15,196  
245 nests/year). Nevertheless, the degree of relative increase remains unclear due to the lack of  
246 pre-exploitation baseline data for Colola (Delgado-Trejo 2016).

247         Synchronous over-exploitation across life-stages and habitats from 1960-1980 likely  
248 contributed to steep population declines by simultaneously decreasing adult survivorship,  
249 hatchling production, and juvenile recruitment (cf. Seminoff & Shanker 2008). Reversing  
250 the negative trends was achieved by concurrent conservation measures across the species'  
251 range in Mexico (Figure 2). Decades of fishing bans and nesting beach protection have  
252 generated a positive feedback loop of increased survivorship and recruitment across life  
253 stages and habitats. Comparisons of trends between nesting and foraging habitats will  
254 provide insight to be evaluated in future research (Van Houtan et al. 2014).

255

### 256 ***Implications for conservation***

257 Our results suggest a robust causal relationship between abundance trends and key events  
258 in conservation and management policies (Figure 4a). BLA's foraging population shows an

259 encouraging trend thanks to decades of conservation measures enacted over broad spatial  
260 scales. Future research agendas will benefit from integrating the effect of climatic  
261 fluctuations, which are drivers of sea turtle population dynamics including hatchling  
262 production, sex determination, juvenile recruitment, foraging success, and the timing and  
263 frequency of reproduction (Patrício et al. 2021). Threats from climate change are more  
264 difficult to mitigate than direct human activity, making tangible conservation efforts  
265 increasingly challenging (Mazaris et al. 2017).

266         While efforts at national, international, and boundary-wide scales are necessary for  
267 effective green turtle conservation, data from foraging areas are crucial for informing local  
268 management (Bjorndal et al. 2005; Broderick et al. 2006). Likewise, multiple forms of  
269 expertise—including locally-grounded collaboration—are essential for creating truly  
270 diverse and inclusive approaches to conservation (cf. Carman & González Carman 2020).  
271 Our results were only possible thanks to long-term, collaborative efforts with the BLA  
272 community, as LEK was indispensable for establishing baseline levels, determining local  
273 recovery targets, and evaluating current population status. Importantly, LEK must be  
274 recognized for its inherent value, and become integral to conservation policy and practice.  
275 As such, any future conservation initiatives—i.e., establishing local habitat protection  
276 areas, by-catch reduction in commercial fisheries, or locally grounded strategies to prevent  
277 unregulated fishing that impact green turtles and their habitats—must be determined by  
278 local communities, with self-determination as the guiding principle of scientific  
279 collaboration (cf. Mawyer & Jacka 2018).

280

## 281 ***Global perspectives and future challenges***

282 Patterns of historical green turtle abundance, decline from overfishing, and growth thanks  
283 to conservation efforts are documented throughout the Pacific Rim, Atlantic, Caribbean,  
284 and Indo-Pacific regions (Broderick et al. 2006; Mazaris et al. 2017). However, climate  
285 change will pose new and increasing challenges (cf. Patrício et al. 2021). LEK-based  
286 approaches will become increasingly relevant as sea turtle populations grow and cease to be  
287 mere conservation targets (cf. Christianen et al. 2021), particularly considering that  
288 conservation conflicts have arisen when management frameworks fail to account for the sea  
289 turtles' cultural importance (cf. Barrios-Garrido et al. 2018). To be successful, future  
290 conservation and policy measures for migratory marine species must integrate international  
291 and basin-wide approaches built upon locally-grounded efforts, integrating the diverse  
292 peoples and worldviews linked to the oceans (Vierros et al. 2020). LEK, accumulated by  
293 people living with and from the sea, is indispensable for comprehending long-term change  
294 and building sustainable futures.

295

## 296 **Acknowledgements and data**

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301 findings of this study include data from CONANP, but restrictions apply to the availability  
302 of these data, which were used under license for the current study, and so are not publicly

303 available. Data are however available from the authors upon reasonable request and with

304 permission of CONANP. Code was implemented in R and is available upon request.

305

306 **Ethical approval:**

307 Research was approved by the Bioethics Committee of the Centro de Investigación

308 Científica y de Educación Superior de Ensenada (Approval Number 2S.3.1).

309

310 **Conflict of interest:**

311 None to declare.

312

313

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Parameter	Pooled estimate	Pooled 95% C.I.	Pooled std. error	P-value	Pooled <i>df</i>	Pooled R <sup>2</sup>
<i>Commercial Fishing (1952-1982; LEK data)</i>						
<i>Model: <math>y \sim \alpha^{(\beta \times x)}</math>; <math>df</math> (complete data) = 29; <math>m = 48</math>; <math>e \sim N(0, \sigma^2)</math> for mean residuals</i>						
$\alpha$	24.271	[19.669 - 28.873]	2.189	< <b>0.01</b>	17.800	
$\beta$	-0.0820	[-0.101 to -0.0628]	0.00928	< <b>0.01</b>	24.060	0.845 [0.692-0.926]
<i>Conservation (1978-2018; LEK and Monitoring data*)</i>						
<i>Model: <math>y \sim \alpha^{(\beta \times x)}</math>; <math>df</math> (complete data) = 38; <math>m = 42</math>; <math>e \sim N(0, \sigma^2)</math> for mean residuals</i>						
$\alpha$	0.0023	[-0.0122 - 0.0168]	0.00703	0.746	25.325	
$\beta$	0.136	[ 0.0760 - 0.196]	0.0286	< <b>0.01</b>	17.376	0.711 [0.364-0.890]

\* LEK values from 1978-1982 were appended to the monitoring dataset to allow for interpolation of values in the temporal gap between LEK and monitoring datasets.

442

443 **Table 1:** Results of Multiple Imputation by Chained Equations (MICE) analysis with non-  
 444 linear model. Bold type indicates significant results at  $\alpha = 0.05$ . Parameter estimates,  
 445 confidence intervals, standard error, and R<sup>2</sup> values were pooled using Rubin's Rules to  
 446 account for uncertainty of the missing data and variance within and between the  $m$  imputed  
 447 models (Dong & Peng, 2013). 95% confidence intervals for R<sup>2</sup> values are shown in  
 448 brackets. Pooled degrees of freedom are included to account for the effects of missing data  
 449 (Supporting Information:4.6) (van Buuren & Groothuis-Oudshoorn, 2011). See also Figure  
 450 5.

451

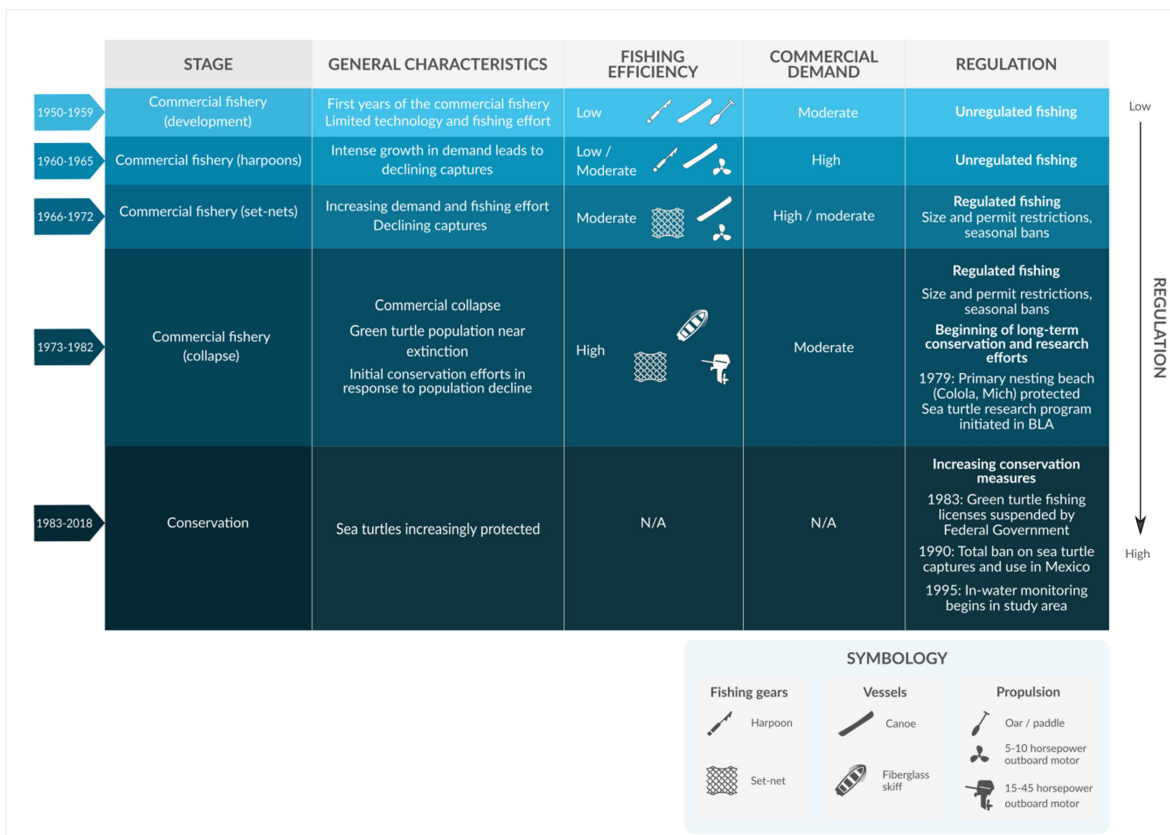


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453

454 **Figure 1:** Map showing the location of the study site at Bahía de los Ángeles, Baja  
455 California, Mexico (orange circle) and Ensenada, Mexico (yellow circle), the primary  
456 market for green turtle catches during the historical commercial fishery. The green square  
457 in the inset map shows the index nesting site in Colola, Michoacán. At the time of this  
458 study, the village of Bahía de los Ángeles had a population of ~500 people, with economic  
459 activity centered on small-scale fishing and seasonal tourism. Map produced in Qgis using  
460 Stamen Terrain Basemap (CC BY 3.0) and data from OpenStreetMap (ODbL).

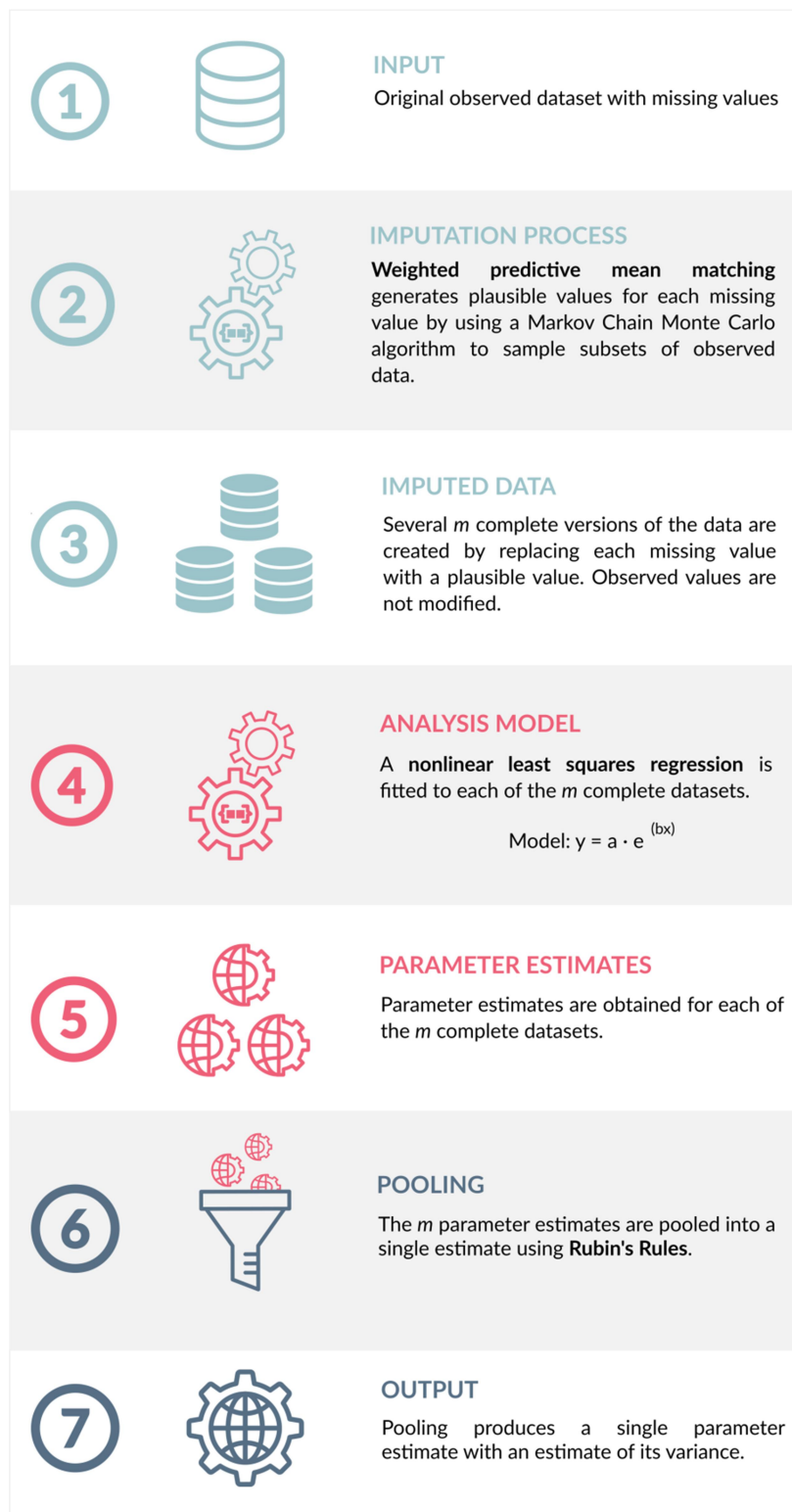
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462

463 **Figure 2:** General chronology of changes in the green turtle fishing and conservation in  
 464 Bahía de los Ángeles. Fishing efficiency increased along with demand between 1960 and  
 465 1980, leading to population collapse. Long-term conservation and research began in the late  
 466 1970s in response to diminishing populations. Fishing permits for green turtles were  
 467 suspended by the Federal Government in 1983 as the fishery collapsed. Regulations and  
 468 conservation measures across green turtle habitats increased from the late 1970s onward,  
 469 reaching full legal protection after the total ban in 1990 (Early-Capistrán et al. 2020;  
 470 Márquez 1996; Seminoff et al. 2008).

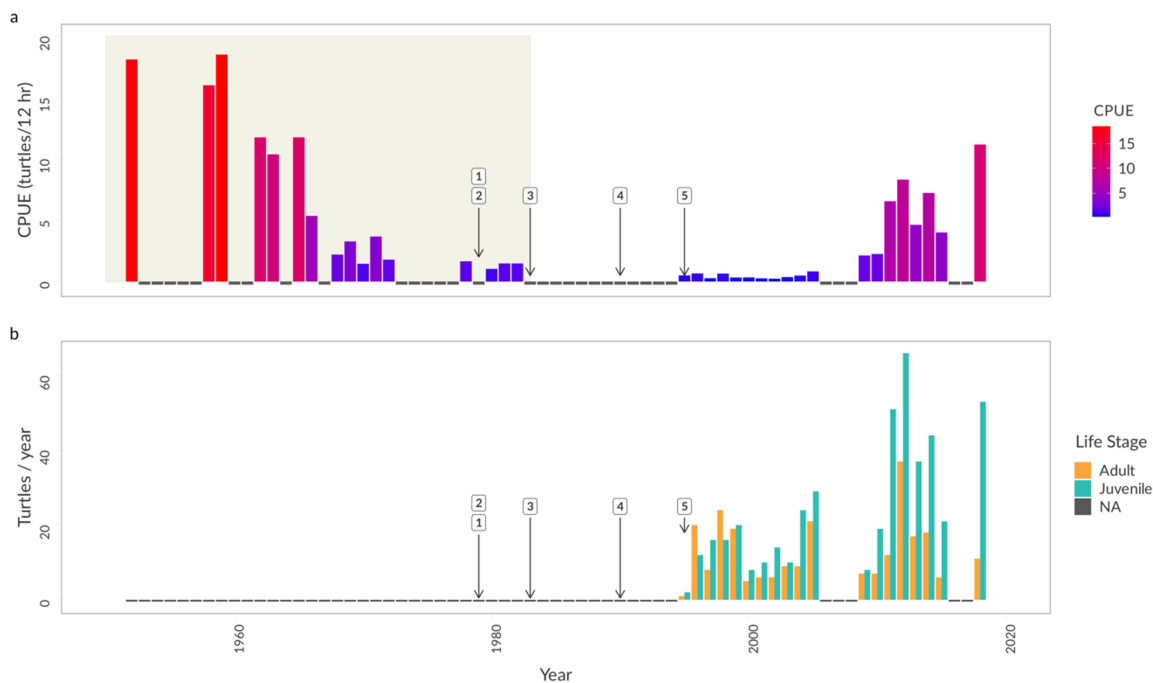
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473 **Figure 3:** Diagram of methodological pipeline used for Multiple Imputation by Chained  
474 Equations (MICE). Models and rulesets are shown in bold.

475



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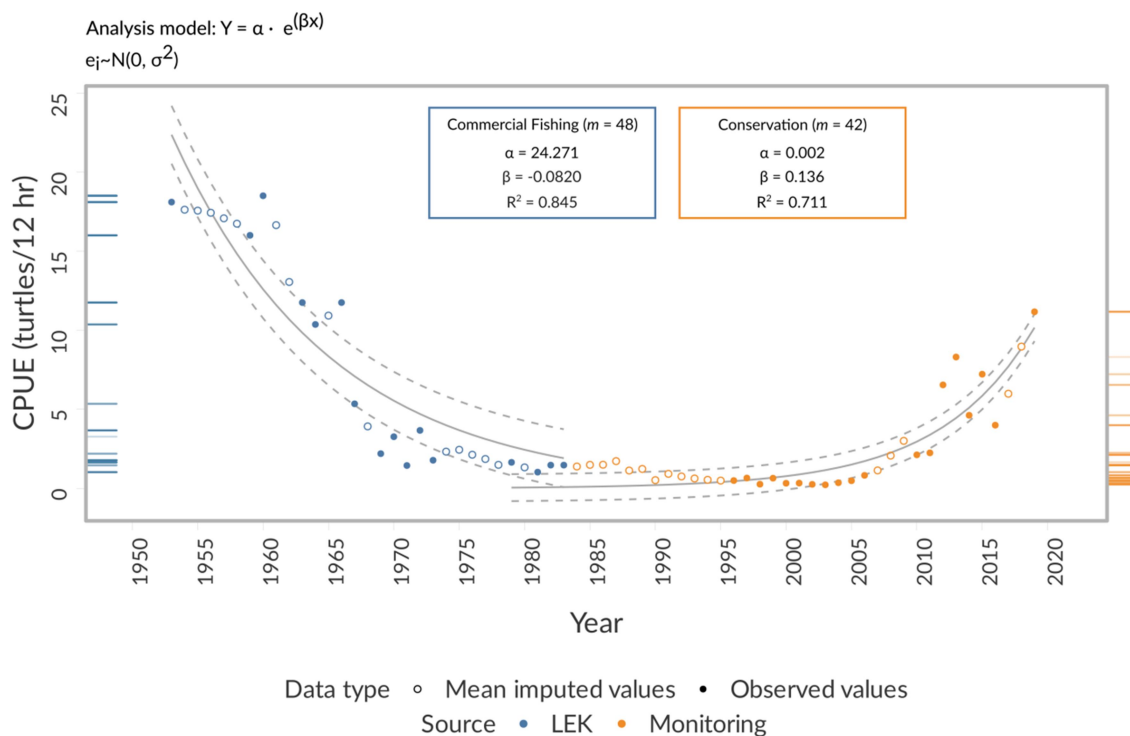
477

478 **Figure 4:** Long-term trends in mean annual catch-per-unit-effort (CPUE) (a) and turtle life  
479 stage distribution (b). Note difference in the y-axis scale. Annotations show key events in  
480 sea turtle conservation and management: (1) start of permanent sea turtle research efforts at  
481 Bahía de los Ángeles; (2) start of nesting beach protection at Colola, Michoacán; (3)  
482 suspension of green turtle fishing permits by the Federal Government; (4) permanent ban  
483 on all sea turtle captures in Mexico; (5) start of in-water monitoring at Bahía de los  
484 Ángeles. Panel (a) shows mean annual CPUE values from 1952-2018. Shaded area shows  
485 data derived from Local Ecological Knowledge. Panel (b) shows total captures per year in  
486 scientific monitoring for adults (Curved Carapace Length, CCL > 82.0 cm) and juveniles  
487 (CCL ≤ 82.0 cm). Size at maturity is based on mean size of nesting females at Colola  
488 (Figueroa et al. cited in Seminoff et al. 2015).

489



490



491

492 **Figure 5:** Trends in catch-per-unit-effort (CPUE) for 1952–2018 using Multiple Imputation  
493 by Chained Equations (MICE). Data points show mean value for each year. Pooled  
494 parameter estimates and  $R^2$  values are shown for the Commercial Fishing phase (1952–  
495 1983; blue box) and the Conservation phase (1978–2018; orange box). Solid trend line  
496 shows pooled predicted values across all  $m$  imputed models, and dashed lines show 95%  
497 Confidence Intervals for the pooled trend line. Standard errors were pooled according to  
498 Rubin's Rules to account for within-model and between-model variance (Dong & Peng  
499 2013) (Supporting Information: 4.5). Marginal rug plots show density distributions of  
500 imputed values for Commercial Fishing (blue rug plot,  $n = 630$ ) and Conservation (orange  
501 rug plot,  $n = 816$ ) (see also Figures S2 and S3). Pooled 95% Confidence Intervals for  
502 parameter estimates and  $R^2$  values are reported in Table 1.

503

## DISCUSIÓN

### **Desarrollo de series de tiempo a partir del CEL**

En el presente trabajo, se analizaron los cambios a largo plazo en la abundancia local de *C. mydas* en el sitio índice de alimentación de Bahía de los Ángeles, partiendo de una serie de tiempo desarrollada mediante la integración de reconstrucciones históricas y datos de monitoreo científico. Para ello, se desarrolló una metodología novedosa para generar estimaciones estandarizadas de CPUE a partir de la sistematización/cuantificación del conocimiento ecológico local usando un proceso iterativo de documentación del CEL a través de la etnografía; procesamiento de datos mediante la codificación, la indexación y la heurística y procesos retroalimentación con análisis estadísticos y modelación matemática (Early-Capistrán et al., 2020). El desarrollo de esta serie de tiempo (1952-1982) permitió (i) establecer una línea base de abundancia previa a la explotación comercial a gran escala y (ii) analizar los cambios en la abundancia durante la pesca comercial y previo al desarrollo del monitoreo científico (Early-Capistrán et al., 2020).

Se encontró que durante la pesquería comercial histórica, particularmente entre 1960 y 1980, hubo un declive pronunciado en la población local de tortuga prieta a causa de la sobreexplotación comercial, debido a la conjunción de la alta demanda de mercado, los incrementos en la eficiencia pesquera y la mejoría de las vías de comunicación que facilitó el acceso a los mercados regionales y nacionales (Early-Capistrán et al., 2020). A inicios de la década de 1980, la tasa de captura de tortuga prieta había disminuido un 95% respecto a los primeros años de la pesquería (~1950), demostrando que la explotación intensiva y

tecnológicamente eficiente con artes como las redes de enmalle —incluso por una pequeña flota artesanal— puede generar impactos severos sobre la abundancia de una especie objetivo de lento crecimiento en un periodo relativamente corto (Longo, 2011; Peckham et al., 2007). El análisis comparativo con registros de capturas anuales totales de la pesquería de *C. mydas* en Bahía de los Ángeles (Márquez citado en Seminoff et al., 2008) y de ajustes de regresión no-lineal, se demostró que los datos estimados a partir del CEL tienen una alta concordancia con los registros de captura total (Lin CCC,  $\rho = 0.726$ ). Asimismo, los modelos ajustados a datos derivados del CEL mostraron un buen ajuste y residuales robustos ( $R^2=0.798$ ;  $e^i \sim N(0, \sigma^2)$ ) (Early-Capistrán et al., 2020).

A través del desarrollo de un marco metodológico que integra herramientas probadas de las ciencias naturales y sociales, se desarrolló una serie de tiempo confiable y estandarizada, que podría integrarse con los datos de monitoreo científico actuales. A través de este proceso, se pudieron establecer estimaciones sólidas a nivel local, comprobar la confiabilidad y robustez de los datos derivados del CEL y desarrollar un marco metodológico innovador y flexible que puede ser adaptado y aplicado en otras especies marinas explotadas a nivel global.

### ***Integración con datos de monitoreo y modelación de cambio a largo plazo***

La serie de tiempo de CPUE estandarizada derivada del CEL (1952-1982) se integró con datos de monitoreo (1995-2018) para generar una serie de tiempo extensa (1952-2018). Mediante el uso de la aproximación MICE, se desarrollaron modelos no-lineales para analizar los cambios en la abundancia local de tortuga prieta en el sitio de estudio, abarcando un

periodo desde los inicios de la pesca comercial hasta el presente. Asimismo, se analizaron los cambios en la composición de tallas a partir de los datos de monitoreo para evaluar los cambios en la abundancia relativa de dos estadios ontogénicos (juveniles y adultos) a lo largo del tiempo. Este proceso permitió (i) analizar los cambios a largo plazo en la abundancia, en relación con los procesos de pesca comercial y conservación; (ii) evaluar el estado de conservación actual en relación con una línea base robusta y (iii) analizar los cambios en la composición de tallas y los procesos de reclutamiento.

Se encontró incremento sustancial de la abundancia de *C. mydas* tras ~40 años de medidas de conservación, ~30 de protección legal plena y ~25 años de monitoreo científico. La población de tortuga prieta en Bahía de los Ángeles disminuyó a una tasa de 8.4% anual durante los años de la pesca comercial, en contraste con un incremento de 4.5% anual tras la implementación de medidas de conservación a finales de la década de 1970. Es decir, el declive poblacional ocurrió 75% más rápido que el aumento. Asimismo, es notable el período prolongado de latencia entre el inicio de las medidas de conservación (1979), el aumento poblacional inicial (~2000) y el aumento poblacional sustancial a partir del 2011 (Early-Capistrán et al., 2021). En la actualidad, si bien hay una clara tendencia al alza, la CPUE máxima en el monitoreo (11.2 tortugas/12hr; 2018) representa ~60% de la CPUE máxima en la pesquería comercial (18.5 tortugas/12hr; 1959) (Early-Capistrán et al., 2021).

El aumento en la abundancia poblacional a partir del 2011 coincide con un incremento en la proporción de tortugas juveniles: los primeros años del monitoreo (1995-2005) se caracterizaron por un leve sesgo juvenil (55.71%), en contraste con un marcado sesgo hacia juveniles (72.22%) entre el 2009 y el 2018 (Early-Capistrán et al., 2021). Estos patrones sugieren que el crecimiento poblacional está impulsado por el reclutamiento preferencial de

tortugas juveniles a este hábitat de forrajeo. Dicho patrón es lo esperado y es consistente con la prolongada historia de vida y el crecimiento lento de las tortugas marinas, que requieren décadas de producción de crías antes que se observe el reclutamiento, primeramente, de tortugas juveniles a las zonas de alimentación (Seminoff & Shanker, 2008).

### **Implicaciones para la conservación**

Los resultados de este trabajo muestran varios procesos contrastantes. Por un lado, se encontró que la pesca comercial, intensiva y con alta eficiencia tecnológica, puede generar impactos severos sobre las poblaciones de tortugas marinas en periodos cortos, incluso en tratándose de una flota pequeña (Peckham et al., 2007). Esto contrasta con los siglos anteriores cuando una pesca de subsistencia con limitaciones tecnológicas y artes de pesca altamente selectivas se podría considerar sustentable (Early-Capistrán et al., 2018). Por otro lado, se observa que las medidas de conservación y monitoreo —protección de hábitats de forrajeo y anidación, prevención y regulación de la captura incidental, prohibición del uso insostenible de carne o huevo y monitoreo sistemático— generan resultados positivos, pero requieren de una implementación en amplias escalas espaciales y temporales. En el caso de la población de Bahía de los Ángeles, el colapso poblacional ocurrió en ~20 años; en contraste, se necesitaron ~40 años de medidas de conservación —implementadas en hábitats de anidación y forrajeo a lo largo del Pacífico mexicano y el Golfo de California— antes de que se observara un crecimiento poblacional sustancial. Estos patrones de declive rápido y recuperación lenta se han observado en diversos taxones marinos longevos (Chaloupka et al., 2008; Lotze et al., 2011; McClenachan et al., 2012).

Es posible que el declive acelerado entre 1960-1980 se haya generado por la sobreexplotación sincrónica en diferentes etapas de vida, ya que la pesquería del Golfo de California surgió casi a la par con la sobreexplotación de huevo en las playas de anidación de Michoacán (Delgado-Trejo & Alvarado Díaz, 2012; Felger et al., 2005). Es probable que estos procesos provocaran un desplome poblacional acelerado al reducir, de manera simultánea, la supervivencia adulta, la producción de crías y el reclutamiento de juveniles (Seminoff & Shanker, 2008). En contraste, la aplicación concurrente de medidas de conservación a lo largo del área de distribución de esta especie en México ha generado un circuito de retroalimentación positiva, al aumentar la sobrevivencia y el reclutamiento entre hábitats y estadios ontogénicos.

### ***Estado de conservación actual en Bahía de los Ángeles***

Con los resultados aquí presentados, resulta indudable que la población de tortuga prieta en Bahía de los Ángeles muestra una tendencia positiva tanto en términos de crecimiento poblacional como en el aumento del reclutamiento de tortugas juveniles. No obstante, la abundancia local sigue por debajo de la línea base histórica: si bien las tendencias de CPUE se encuentran alrededor de los niveles de la década de 1960, aún se encuentra ~40% por debajo del nivel de los primeros años de la pesquería. Cabe resaltar que esto coincide con el consenso entre los pescadores expertos de mayor edad, quienes concuerdan que a pesar de que la tortuga prieta es abundante en la actualidad y, aunque sus poblaciones han crecido, sigue por debajo de los niveles que recuerdan haber observado cuando eran arponeros en su juventud (Early-Capistrán et al., 2021). Estos resultados sugieren que, si bien ya se están observando los resultados positivos de las décadas de conservación y monitoreo, es

fundamental continuar con estas medidas para que la población recupere a sus niveles de abundancia histórica.

### **El CEL y los esfuerzos futuros de conservación**

La complejidad espaciotemporal de la historia de vida de la tortuga prieta conlleva la necesidad de implementar medidas de conservación a largo plazo en escalas regionales, nacionales e internacionales (Broderick et al., 2006; Mazaris et al., 2017). Los esfuerzos futuros de conservación de especies longevas y altamente migratorias, como las tortugas marinas, requieren de esfuerzos coordinados a partir de casos exitosos a nivel local (Carman & González Carman, 2020; Vierros et al., 2020). El caso de la tortuga prieta destaca como un ejemplo de participación colectiva ejercida durante décadas, mediante la cooperación de instancias de gobierno, organizaciones de la sociedad civil y decenas de grupos comunitarios de monitoreo a lo largo del Golfo de California y el Pacífico mexicano (Koch, 2013; Márquez, 1996; Seminoff et al., 2008). Los esfuerzos futuros se podrán ver fortalecidos al integrar la diversidad de perspectivas, conocimientos y habilidad de la comunidad científica y de las comunidades locales, promoviendo un uso emancipador del conocimiento científico (Carman & González Carman, 2020; Game et al., 2015; Lee et al., 2018). Las aproximaciones colaborativas y transdisciplinarias —que incluyen la integración del conocimiento científico y el CEL— serán fundamentales para asegurar procesos de conservación equitativos e incluyentes (cf. Peacock et al., 2020).

La participación colectiva y la investigación transdisciplinaria —que incorporan las múltiples formas de conocimiento de todos los sectores involucrados y se sustentan en las

condiciones locales— son esenciales para el abordaje de los problemas de conservación complejos (Brown, 2010; Carman & González Carman, 2020). Los esfuerzos futuros deberán reconocer el valor inherente del CEL e integrarlo plenamente en los planes de conservación y manejo, dentro de un marco de colaboración equitativa entre la comunidad y la investigación científica. A la vez, cualquier medida a implementarse —por ejemplo, el establecimiento de hábitats protegidos a nivel local, medidas para prevenir la captura incidental o prevenir la pesca no-regulada con impactos potenciales en las tortugas marinas o sus hábitats— deberán ser determinados por la comunidad, partiendo de un principio de autodeterminación, dentro de un esquema pluralista en el que se promueva el acceso a la información científica (cf. Mawyer & Jacka, 2018). Esto puede lograrse mediante un esquema de investigación-acción-participación encaminado al diálogo epistémico (cf. Carman & González Carman, 2020; Colmenares E., 2012; Mistry & Berardi, 2016). Asimismo, estos esfuerzos colaborativos deberán buscar facilitar el entendimiento compartido y asegurar el beneficio mutuo (cf. Bennett et al., 2017).

El uso del CEL cobra particular importancia ante los aumentos actuales de las poblaciones de *C. mydas* en diversas regiones del mundo, sobre todo considerando los huecos de información y la falta de datos de línea base en la mayoría de las mismas (Broderick et al., 2006; Chaloupka et al., 2008; Mazaris et al., 2017). Asimismo, es importante resaltar que gran parte de los programas de conservación a nivel mundial han fracasado o generado conflictos al no considerar la importancia sociocultural de las tortugas marinas para las comunidades locales (Barrios-Garrido et al., 2017; Campbell, 2007; Mohd Jani et al., 2020). A la par, debe tomarse en cuenta el papel fundamental de los agentes locales como motores de la conservación, el conocimiento y la identidad de los paisajes culturales marinos (cf.



Hamada, 2020; Mawyer & Jacka, 2018). Para ello, es fundamental adoptar una aproximación crítica, asegurando que el uso del CEL no se desarrolle en un marco que cimente procesos que conlleven a la exclusión, marginación o despojo de las poblaciones que lo poseen (Bennett et al., 2015; Trisos et al., 2021; Vertovec & Posey, 2004). Esto ha sucedido históricamente en contextos de conservación de especies en riesgo, incluyendo las tortugas marinas, y continúa sucediendo ante procesos como el desarrollo de la economía azul (Bennett et al., 2015; Campbell, 2007; Early Capistrán, 2010; Longo et al., 2015; Shanker & Kutty, 2005).

Conforme las poblaciones de tortuga prieta pasan de ser escasas a aumentar y recuperar sus papeles funcionales dentro de los ecosistemas marinos, se desarrollarán debates cada vez más complejos (Christianen et al., 2021; Heithaus et al., 2014; Lal et al., 2010). Este nuevo panorama de complejidad creciente requerirá, cada vez más, de aproximaciones transdisciplinarias y colaboraciones sólidas entre todos los sectores involucrados (Brown et al., 2010; Carman & González Carman, 2020). Ante este escenario, las aproximaciones presentadas en este trabajo podrán ser adaptadas para la reconstrucción y evaluación de las tendencias poblacionales de especies longevas como las tortugas marinas a largo plazo en diferentes regiones del mundo y, de manera importante, para las zonas de forrajeo para las cuáles la información técnica disponible es más escasa a nivel mundial (Early-Capistrán et al., 2020; Seminoff & Shanker, 2008).

## Visión a futuro

En el presente trabajo se demuestra, en conjunto con trabajos previos, la viabilidad del CEL no solo para evaluar la población local de *C. mydas*, sino para generar herramientas y aproximaciones que aporten a la resolución de los problemas complejos implícitos en la conservación de las especies marinas. En el caso de la tortuga prieta, la compleja historia de vida, en conjunto con la larga trayectoria de explotación e importancia sociocultural, la convierten en un importante estudio de caso a través de la cual se han podido desarrollar herramientas relevantes a otras regiones y especies. En este trabajo, se aporta una aproximación transdisciplinaria que integra métodos robustos de la etnografía y las matemáticas que permiten que el CEL se integre a la modelación ecológica con el rigor necesario para la evaluación de las poblaciones de especies silvestres en riesgo.

En el caso de *C. mydas*, su patrón de dinámica poblacional observado en Bahía de los Ángeles es similar a otras regiones de la cuenca del Pacífico, el Caribe, la región Indo-Pacífico y el Atlántico, donde se ha observado un declive marcado debido a la sobreexplotación, seguido de la implementación de medidas de conservación y diferentes grados de incremento poblacional después de años o décadas (Chaloupka et al., 2008; Lotze et al., 2011; Mazaris et al., 2017). El marco metodológico presentado en el artículo de *PeerJ* (Early-Capistrán et al., 2020) podría modificarse para desarrollar estimaciones robustas y líneas base en aquellas poblaciones donde no existen datos a largo plazo, facilitando el conjunto de evaluaciones a nivel regional necesarias para evaluar el estado de conservación a nivel global de esta especie cosmopolita (Broderick et al., 2006; Seminoff et al., 2015; Wallace et al., 2010).

La generación de información a largo plazo de las tendencias poblacionales en un hábitat de forrajeo constituye un aporte importante a la investigación básica de *C. mydas*, ya que la falta de datos sobre las poblaciones en sus hábitats de alimentación —que incluyen tortugas adultas y juveniles de ambos sexos— ha limitado el entendimiento de los parámetros demográficos y la conectividad entre hábitats y estadios de vida (Seminoff et al., 2015; Van Houtan et al., 2014). La disponibilidad de datos de abundancia a largo plazo en sitio índice de forrajeo podrá contribuir, mediante análisis conjuntos y comparativos con los sitios de anidación, al desarrollo de parámetros fundamentales como la edad de madurez sexual y la duración de etapas ontogénicas (Casale & Heppell, 2016; Seminoff & Shanker, 2008). Asimismo, en investigaciones futuras, podría contribuir al entendimiento del impacto de los fenómenos climáticos en las tendencias de abundancia y la conectividad demográfica y entre hábitats (Patrício et al., 2021; Seminoff et al., 2015). Estas investigaciones potenciales serían de gran utilidad para avanzar en un entendimiento holístico de los diversos aspectos de la ecología y biología de las tortugas marinas. Dada que esta constituye —hasta donde se sabe— la serie de tiempo más extensa disponible para un hábitat de alimentación de *C. mydas*, brinda posibilidades importantes y únicas para el entendimiento de las dinámicas de esta especie.

## CONCLUSIONES

La trayectoria de la abundancia poblacional de *C. mydas* coincide con la de varias poblaciones alrededor del mundo: cambio lento a lo largo de siglos, seguido por declives acelerados (Early-Capistrán et al., 2018; Jackson et al., 2001; Lotze & Worm, 2009). En algunos casos, las medidas de conservación y protección han llevado a incrementos o recuperaciones aparentes que deben ser evaluados respecto a una línea base robusta (Lotze et al., 2011; Mazaris et al., 2017). No obstante, en la mayoría de los casos, no existen datos confiables a largo plazo que permitan hacer dichas comparaciones (McClenachan et al., 2016; Pauly, 1995; Sáenz-Arroyo et al., 2005). La península de Baja California ofrece un estudio de caso interesante donde, debido a la larga trayectoria de uso sustentable y el declive poblacional relativamente reciente, el nivel de abundancia previo a la explotación a gran escala se puede registrar a partir del conocimiento de los pescadores de mayor edad (Early-Capistrán et al., 2018).

En este trabajo, se presentó una metodología innovadora para generar estimaciones confiables y estadísticamente robustos a partir del conocimiento ecológico local (Early-Capistrán et al., 2020). Estas estimaciones estandarizadas se pudieron integrar directamente con datos de monitoreo ecológico para generar —hasta donde se sabe— la serie de tiempo más extensa disponible para un hábitat de forrajeo de tortuga marina a nivel mundial (1952-2018) (Early-Capistrán et al., 2021). Esta serie de tiempo permitió evaluar los cambios en abundancia a largo plazo y evaluar el estado de la población a nivel local respecto a una línea base histórica.

A grandes rasgos, se encontró que la pesca comercial intensiva y tecnológicamente eficiente generó un declive exponencial en un periodo corto (1960-1980). Cabe destacar que este declive no ocurrió debido al consumo local, sino (i) a la existencia de una demanda elevada en los centros urbanos, surtida por una pesquería con escasa regulación y (ii) la facilidad para surtir la demanda de tortugas mediante incrementos en la eficiencia pesquera y rutas de acceso a los mercados (Early-Capistrán et al., 2018, 2020). El aumento paulatino de las medidas de conservación ante la reducción de las poblaciones —comenzando con la protección de playas de anidación en Michoacán en 1979, seguido de la suspensión de licencias de captura de tortuga prieta en 1983, la veda total en 1990 y el inicio de programas de monitoreo en el agua en BLA 1995— ha llevado a un incremento paulatino en la abundancia poblacional en Bahía de los Ángeles, impulsado por el reclutamiento de tortugas juveniles (Early-Capistrán et al., 2021; Márquez, 1996; Seminoff et al., 2008).

Los resultados del presente trabajo indican que, en la actualidad, la población local de tortuga prieta se encuentra a ~60% del nivel histórico de línea base. Los resultados de este trabajo sugieren que las medidas de conservación —protección y monitoreo de hábitats de anidación y forrajeo, prevención y regulación de la captura incidental y prohibición del uso insostenible de carne o huevo— han generado cambios positivos. No obstante, estas medidas deben implementarse durante décadas; en este caso, el aumento poblacional inicial es el resultado de más de 40 años de esfuerzos de protección. Asimismo, requieren de esfuerzo locales replicados a lo largo de la distribución de la población. En este caso, han existido décadas de esfuerzos coordinados entre instancias de gobierno (de manera notable, la Comisión Nacional de Áreas Naturales Protegidas), de la sociedad civil (de manera destacada, el Grupo Tortuguero de las Californias A.C. y decenas de grupos locales de

monitoreo a lo largo del Pacífico mexicano y el Golfo de California) y de instancias académicas (Koch, 2013; Márquez, 1996; Seminoff et al., 2008).

Amén de los resultados que sugieren una recuperación incipiente y endeble, es notable que la tasa de declive durante la pesquería comercial (8.4% anual) es un 75% mayor que la tasa de incremento a partir de la implementación de medidas de conservación (4.8% anual). La compleja historia de vida, el crecimiento lento y la madurez sexual tardía de *C. mydas* la hace una especie vulnerable a la explotación humana en todas sus etapas de vida (Seminoff et al., 2015; Seminoff & Shanker, 2008), por lo que es fundamental continuar con las medidas de conservación para que esta población recupere su abundancia histórica. Este patrón se ha observado en otras poblaciones de tortugas marinas que, si bien han mostrado aumentos relativos, en pocos casos han alcanzado niveles históricos (Chaloupka et al., 2008; Lotze et al., 2011; Mazaris et al., 2017).

El cambio climático traerá consigo nuevos retos para las tortugas marinas (Maurer et al., 2021; Mazaris et al., 2017; Patrício et al., 2021). El desarrollo de medidas de conservación y manejo de esta y otros taxones marinos altamente migratorios podría beneficiarse del aprendizaje que brinda el caso de la tortuga prieta: se requerirán de esfuerzos coordinados a nivel local, regional, cuenca oceánica y global con perspectivas a largo plazo. Estos esfuerzos deberán, también, nutrirse del conocimiento acumulado por las poblaciones y que viven con y del océano, partiendo de un principio de autodeterminación y colaboración equitativa (cf. Vierros et al., 2020). La vinculación del conocimiento ecológico local con el conocimiento científico es una manera de aprovechar la capacidad colectiva de las comunidades locales y la comunidad científica (Game et al., 2015). En este sentido, el

conocimiento ecológico local será fundamental para entender los cambios a largo plazo y crear futuros sustentables.

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